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Ecosystem Recovery Planning for Listed Salmon: An Integrated Assessment Approach for Salmon Habitat

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EXECUTIVE SUMMARY

The Endangered Species Act (ESA) requires that a recovery plan be developed for all listed threatened and endangered species. For Pacific Salmon, this includes 26 Evolutionarily Significant Units (ESUs) of 6 species of salmon and steelhead trout distributed among 9 geographical areas along the West Coast. Several factors associated with harvest, hatchery, hydropower, and habitat have been identified as important to understanding the decline in salmon and should be addressed in a successful recovery plan.

Previous documents have been written to provide guidance for recovery planning (McElhany et al. 2000, NMFS 1992, 2000); however, specific guidance on how to implement the habitat portion of a recovery plan is not adequately addressed. This document was written to supplement prior guidance documents with information specific to habitat recovery planning. It also addresses several issues related to hydropower, including altered flow regimes and hydrology. It is not intended that existing habitat recovery planning efforts (e.g., at the local watershed level) should be abandoned in favor of methods discussed here, but rather these methods should provide a context for understanding the principles of habitat recovery planning and suggest tools for conducting these assessments. Additionally, following this process may suggest areas where further efforts should be focused. Audiences that may benefit from this document include Technical Recovery Teams (TRTs) assigned to each of the geographical planning areas, local watershed planning groups, and National Marine Fisheries Service (NMFS) personnel. Although not addressed in this document, information provided by following the

guidelines in this document may be beneficial in other ways, such as increased compliance with environmental regulations (including other portions of the ESA).

An Assessment Approach for Habitat Recovery Planning

The conceptual approach suggested here for habitat recovery planning is based on an ecosystem design. Salmon are adapted to local environmental conditions (including associated temporal and spatial variability). Landscape factors (e.g., geomorphology, climate) and land uses (e.g., residential, agricultural) affect ecosystem functions and processes (e.g., hydrology, sediment transport, riparian function), which shape the local habitat characteristics on which salmon rely. Thus an approach to habitat recovery that focuses on restoring (or preserving) ecosystem processes should provide quality salmon habitat over the long term. An advantage of this approach is that it is not species-specific and could provide benefits to an entire ecosystem. This may ease or eliminate conflicting management needs.

Recovery planning should also take into account the differences associated with ecoregions (e.g., coastal forests, western forested mountains, western deserts). Any given ESU may occur within one or more ecoregions, and differences associated with each ecoregion (e.g., climate, lithology, topography) may affect the relative importance of ecosystem processes affecting salmon. Assessments should investigate all processes; however, the importance of each will vary by region.

Ideally, ESU-wide assessments should be conducted prior to conducting watershed-level assessments because they can suggest areas that need further attention. Results from both assessments can be used to prioritize restoration actions. However, local efforts already underway may be a product of management needs and what information is already available, and should not be abandoned. New information gained from assessments can supplement ongoing activities and suggest appropriate future actions. Successful recovery plans should be implemented as strategies that identify goals, existing information and gaps, and what information is needed to fill the gaps and attain the goals. These strategies will likely need to be revised as new information is gained from ongoing assessments.

Habitat Analyses for Phase I Recovery Planning: Setting Recovery Goals

The first phase of habitat recovery planning is associated with broad-scale (e.g., ESU-wide and watershed-scale) questions. Specifically, 1) How might habitat changes have altered the abundance, productivity, spatial structure and diversity of individual populations? and 2) What scenarios of habitat characteristics would support a viable ESU? These questions can be addressed by assessments and analyses conducted at several levels: 1) ESU-wide analyses can provide an overall understanding of broad-scale patterns of land use and habitat conditions, and relate these to the four viable salmonid population (VSP) parameters (salmon abundance, productivity, life history diversity, and spatial structure), and 2) watershed-level analyses can elucidate such patterns specific to each watershed. Results of these analyses can be used to set biological delisting criteria (recovery goals) for each of the salmon ESUs.

This approach is meant to provide information about a broad geographic area in a relatively short period of time using existing data. At the ESU scale, using a consistent methodology (e.g., geographic information systems (GIS) and remote sensing) enables comparisons of results from multiple geographic areas and ecoregions within ESUs. Correlation analyses can be performed on geo-referenced data to determine the relationship among natural landscape attributes, land uses, and effects on salmon (e.g., Salmonid Watershed Analysis Model). Comparisons between historic (ideal) and current habitat conditions can help assess potential productivities and capacities of salmon populations.

At the watershed level, similar analyses can be conducted to ascertain relationships among landscape attributes, land uses, and salmon viability. Existing tools include GIS analysis that predicts the amount of habitat potentially available to salmonids and the Ecosystem Diagnosis and Treatment model. Both methods use data relevant to a particular watershed; GIS analyses rely on simple relationships (e.g., habitat-productivity) derived from empirical data, whereas EDT relies on expert opinion, existing survival estimates, and biological rules in a complex model.

Information gained from this type of analysis includes current and historical estimates of habitat patterns and associated use by salmon. These assessments should produce patterns of habitat alteration over a broad geographic area and highlight areas where such change may have most greatly affected salmon viability. Land use practices associated with habitat losses can be correlated with salmon abundance to identify potential causes for decline. Analyses should also define areas of high habitat quality sufficient to sustain viable populations. Results can be used to identify criteria required to sustain a viable ESU and which populations in the ESU are necessary for an ESU to be viable. Results can also suggest areas where further or more detailed research should be conducted at the watershed level to gain a more complete understanding of factors affecting salmon viability.

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Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions

The second phase of recovery planning is associated with several questions that are most appropriately investigated at the watershed level. Specifically, 1) Where have habitat-forming processes and ecosystem functions been impaired? and 2) Where has biological integrity been degraded? These assessments target areas that were identified during ESU-level assessments, but can also be conducted independent of ESU-level assessments (e.g., existing and ongoing activities). The primary purpose of these assessments is to collect and analyze data necessary to identify restoration actions within each watershed that may be necessary for recovery of salmon. Actions that focus on restoring or preserving ecosystem functions or processes (e.g., sediment transport, hydrology, water quality, and habitat connectivity) have the best likelihood for successful long-term recovery of salmon.

The methods associated with this approach include inventories and assessments that identify disrupted ecosystem processes and impaired biological integrity. Analyses are conducted in various watersheds within the geographical area comprised by an ESU. Some watersheds span more than one ecoregion (e.g., coastal forests, western forested mountains, western deserts); thus important processes may differ among regions. Inventories that identify altered ecosystem processes investigate distributed (i.e., widespread non-point such as sediment supply inventories) and reach-level (e.g., floodplain and riparian characterization) watershed processes, as well as other ecosystem functions not easily described by rates or levels (e.g., barrier and flow-diversion inventories). Assessments that identify impaired biological integrity (e.g., B-IBI, multivariate model analyses) can identify locations where habitat degradation may be altering biological communities and which ecosystem processes have been disrupted.

These analyses in combination aim to identify the natural landscape processes active in a watershed, the effects of land use on natural processes, and the causal relationships between land use and habitat conditions. Specific results include identification of locations where stream segments, reaches, or subwatersheds are most impaired, and possible causes of impairment. Additionally, assessments can associate seasonal fish use and survival with habitat type (e.g., habitat use varies among life history stages). From these assessments, a list of recovery actions can be prepared for each watershed in an entire ESU. Ongoing assessments can also help to refine the quality of data and modeling tools used to predict responses to restoration actions as recovery plans are reviewed and modified.

Prioritizing Potential Restoration Actions within Watersheds

Previous assessments conducted at the ESU and watershed levels can be used to prioritize restoration actions necessary for a habitat recovery plan. Information from assessments can also supplement existing restoration activities. Activities at this level are generally referred to as part of Phase II recovery planning.

Several major habitat restoration techniques exist: 1) habitat reconnection, 2) road improvement, 3) riparian restoration, 4) instream habitat restoration, and 5) nutrient enrichment. These and other techniques should be evaluated to determine which would be most effective for a given habitat recovery plan. By following a hierarchical strategy for prioritizing site-specific restoration activities within each watershed in an ESU, a recovery plan has the greatest chance of success. Highly prioritized actions should have a high probability of success, low variability among projects (i.e., consistency of results), a relatively quick response time, and a long duration of results. Cost and time required to implement actions as well as immediate management needs should also be considered during prioritization. Generally, preservation of high quality habitat should take highest priority. Next, restoration of habitat-forming processes or reconnection of isolated habitat should be considered. Finally or in conjunction with higher priority actions, there are site-specific restoration activities. However, site-specific habitat enhancement strategies may be useful for short-term habitat improvements for endangered species.

Following this process should allow salmon recovery to proceed with interim restoration actions that have the highest probability of success (and lowest cost) while assessments and analyses at the ESU and watershed levels define and refine conservation requirements.

Issues of Scale in Habitat Recovery Planning

Stream systems can be thought of as continuous and spatially hierarchical (e.g., river continuum concept), and are subject to processes occurring at different temporal scales (e.g., geologic events vs. habitat-forming events). Additionally, streams are heterogeneous; the conditions found at any given location are a product of processes occurring at various spatial and temporal scales. Understanding how ecosystem processes interact over various spatial and temporal scales should be incorporated into habitat recovery planning.

Spatial and temporal scales of investigation affect interpretation of results. Often, data used to make predictions or set goals is collected at a scale different than the targeted results, and extrapolation from one scale to the other may not be warranted. However, combining information collected at various scales can provide a broader understanding of both the processes impaired as well as recovery potential, and can result in more robust predictions. For example, consider how spatial scale can be used to accomplish recovery goals. Broad-scale analyses can suggest areas where further analyses need to be conducted at a finer scale. Broad-scale analyses identify patterns whereas fine-scale analyses elucidate impaired processes. Thus, conducting a multiscale approach to data collection and analyses can suggest areas where the most effort should be focused.

Managing Uncertainty in Habitat Recovery Planning

Acknowledging and describing uncertainty associated with assessments and analyses can increase the effectiveness of recovery planning. This process elucidates the full range of possible outcomes, and the probability of seeing each of these outcomes. Knowing where uncertainties exist allows managers to develop plans with acceptable risk (i.e., plans where the benefits of an action outweigh its costs).

By recognizing where uncertainty exists, areas that need further clarification (e.g., more data, additional expert opinion, better model performance) can be identified. To do this, estimates of the magnitude of uncertainty in each of five types of uncertainties must be generated: 1) prediction, 2) parameter, 3) model, 4) measurement, and 5) natural stochastic variation. Identification of which types of uncertainty are likely to have the largest effect on predictions can suggest areas where improvements in information will be most beneficial.

Often, decisions need to be made before adequate data are available. Provided that uncertainties are identified, several established methods can be used to make decisions based on the best available information. These methods aid in prioritization of actions, and are preferred over methods that rely on guesswork, biased data, or data collected at inappropriate scales. Final outcomes chosen must be robust to each of the types of uncertainty identified. These decision strategies should assist in creating sound plans in the interim, and can be re-evaluated as new information is obtained.

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INTRODUCTION

One of the main purposes of the Endangered Species Act (ESA) of 1973 is "to provide a means whereby the ecosystems upon which endangered species and threatened species depend may be conserved." The ESA consequently requires the development and implementation of recovery plans in order to affect the conservation of listed species, and details that recovery plans must include:

1) a description of such site-specific actions as may be necessary to achieve the plan's goal for conservation and survival of the species;
2) objective, measurable criteria which, when met, would result in a determination that the species be removed from the list; and
3) estimates of the time and cost required to carry out those measures needed to achieve the plan's goal and to take the intermediate steps toward that goal.

For ESA-listed salmon in the western United States, this requirement is no small task, as salmon habitat is ubiquitous and the actions required to protect or restore the ecosystems on which salmon depend are in conflict with most land uses in the region.

The ESA provides little guidance concerning the content of recovery plans for individual species. Therefore, the National Marine Fisheries Service (NMFS) provides additional scientific guidance on setting recovery goals for evolutionarily significant units of salmon (ESUs) and the populations within them (McElhany et al. 2000), based on the concept of viable salmonid populations (VSPs). (An ESU, equivalent to a "distinct population segment" under the ESA, is "a population or group of populations that are 1) substantially reproductively isolated from other populations, and 2) contribute substantially to the ecological or genetic diversity of the biological species" [Myers et al. 1998]). For each ESU, recovery goals generally are concerned with identifying how many and which independent populations are necessary for ESU viability (McElhany et al. 2000). McElhany et al. (2000) also identify four types of recovery goals that must be met for each population within an ESU of listed salmon: abundance, productivity, spatial structure, and diversity. However, the VSP guidance did not address how to identify specific recovery actions for harvest, hydropower, hatcheries, or habitat that were necessary to achieve ESU or population viability.

In addition to the VSP guidance, NMFS provides guidance to the Technical Recovery Teams (TRTs), which are tasked with developing the technical aspects of a recovery plan for each ESU of listed salmon (NMFS 2000, referred to as the TRT Guidance Document). This guidance document identifies two phases of recovery planning: Phase I identifies the recovery goals (i.e., criteria that must be met for delisting), and Phase II identifies restoration actions that will be necessary for recovery. In this guidance, the habitat elements of the TRT work program are mainly included in Phase II planning, and are identified as:

1) describe fish-habitat productivity relationships,
2) identify factors for decline and limiting factors, and
3) identify early actions for recovery.

The TRT Guidance Document goes on to indicate that characterizing the habitat/fish productivity relationship includes assessing the spatial distribution of fish abundance for each population in the ESU, associating fish abundance with habitat characteristics, and identifying human factors that have the greatest impact on key freshwater and marine habitat. However, it does not specify appropriate spatial scales or resolution levels of data analyses. Moreover, it does not clearly elucidate the questions that such analyses are intended to answer, especially as they relate the population goals for diversity and spatial structure. The TRT Guidance Document also stops short of specific questions for identifying limiting factors and identifying habitat recovery actions.

As neither the VSP or TRT Guidance Documents addressed how to identify specific habitat actions that would support salmon recovery, the Watershed Program at the Northwest Fisheries Science Center was asked to develop technical guidance for the habitat restoration elements of salmon recovery plans. In response to this request, this report describes an approach to developing the habitat elements of a recovery plan for salmon listed under the ESA, including analyses to assist in both Phase I and Phase II planning (Table 1-1). We begin with a conceptual framework for understanding relationships among land uses, watershed functions, habitat conditions, and biota ("An Assessment Approach for Habitat Recovery Planning" section). The conceptual framework relies on principles of watershed and ecosystem management, and organizes the habitat-related questions that each recovery plan should attempt to answer. These questions first address how habitat changes might have affected abundance, productivity, spatial structure, and diversity of salmon populations within an ESU (questions relevant to Phase I recovery planning, or setting the recovery goals). Some of these questions are relevant at the spatial scale of the entire ESU, whereas other questions must be answered for each population within an ESU. A second group of questions addresses causes of habitat change, which provides the basis for identifying restoration actions that are necessary to recover the ecosystem upon which salmon depend (Phase II recovery planning).

After listing the important questions, we provide a brief overview of methodologies that are appropriate for answering each question. These methods are designed to assist in setting recovery goals as well as in identifying restoration actions. In the "Habitat Analyses for Phase I Recovery Planning: Setting Recovery Goals" section, we discuss analyses that should be of use in setting recovery goals. We anticipate that TRTs and their member agencies will largely be responsible for these analyses. In the "Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions" section, we describe how ESU-level assessments can be used to create a broad understanding of habitat issues affecting salmon populations across an ESU. These types of assessments will also typically be under the purview of TRTs, and only provide a general sense of the types and magnitudes of land use alterations affecting the performance of different salmon populations. In the "Prioritizing Potential Restoration Actions Within Watersheds" section, we describe more detailed assessments to be conducted within individual watersheds that identify causes of habitat loss or degradation, as well as changes in habitat or biological conditions. These analyses will most likely be the responsibility of local watershed groups. Data that identify causes of habitat change can be used to develop restoration and protection actions, whereas data describing habitat condition or biological status of streams can be used to prioritize recovery actions and monitor progress toward recovery.

Finally, new information acquired over the next several years will continually modify our understanding of how ecosystems have been altered, and of how those alterations have affected salmon populations. We can be certain that much of what we learn will change our prioritization of recovery actions, and that our recovery plans will need to be updated as new information comes in. In the "Issues of Scale in Habitat Recovery Planning" section, we consider how new information can be incorporated into recovery plans and prioritized lists of habitat recovery actions. In the "Managing Uncertainty in Salmon Habitat Recovery Planning" section, we discuss how uncertainty may affect planning decisions and provide guidance and examples for identifying and quantifying types of uncertainty. Then in Appendix A and Appendix B, we discuss issues of scale in watershed and habitat analysis as well as provide concrete examples of how different questions can be answered using the methodologies we describe in this document.

We recognize that many existing assessment methodologies already incorporate many aspects of the guidance that we provide here (e.g., Moore 1997, Quigley and Arbelbide 1997, SWC 1998, OWEB 1999a, JNRC 2001), and we support existing approaches that favor restoration of ecosystem processes and functions. We do not intend that these methods be abandoned in favor of a redesigned assessment, but we believe that this guidance may be useful to existing TRTs and local watershed groups in clarifying the specific purposes and methods of assessment within their existing approaches. In addition, there are many TRTs yet to be formed and many local watershed groups that have not yet formulated an approach and methodology for recovery planning. These groups are the primary audience for this document, and the purpose of this report is to help them identify specific recovery planning questions, assemble appropriate methods for answering those questions, and utilize that assessment information to identify and prioritize ecosystem recovery actions.

We stress that in this report we focus on habitat analyses that can help in identifying specific ecosystem restoration actions that are necessary to recover ecosystems that support salmon, as well as in setting population recovery goals. However, there are many other aspects of recovery planning that we do not address in this report. First, we do not address issues of large hydropower systems in the Columbia River basin, exotic species impacts, or harvest and hatchery practices. We do, however, address flow diversions and alteration of hydrologic regimes by individual dams and diversions as one component of ecosystem assessments. We also do not specifically address certain regulatory statutes that may be considered programmatic elements of a recovery plan. Regulations such as water quality standards, forest practices rules, the Northwest Forest Plan, and local growth management ordinances should serve as ecosystem protection actions at a minimum, and may serve as passive recovery actions in the best case (e.g., where substantial riparian buffers allow natural recovery riparian processes and functions). A companion document addresses many of the issues associated with incorporating such actions into a final recovery plan.

Table 1-1. Topics covered in this report, and section locations for each topic.

Topic	Section location in this report
Conceptual framework for ecosystem recovery planning	"An assessment Approach for Habitat Recovery Planning"
Phase I planning: habitat analyses to assist in setting population recovery goals	"Habitat Analyses for Phase I Recovery Planning: Setting Recovery Goals"
Phase II planning: habitat analyses for identifying and prioritizing recovery actions	"Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions" "Prioritizing Potential Restoration Actions Within Watersheds"
Decision making and uncertainty	"Managing Uncertainty in Salmon Habitat Recovery Planning"
Issues of scale	"Issues of Scale in Habitat Recovery Planning" Appendix A
Example of watershed-level analyses for identifying and prioritizing recovery actions (Phase II planning)	Appendix B

AN ASSESSMENT APPROACH FOR HABITAT RECOVERY PLANNING

Timothy J. Beechie

In this section we briefly describe an approach to understanding ecosystem functions and habitat change, as well as the scientific and practical reasons for choosing this approach. For the purposes of this report, the ecosystem we discuss includes freshwater and delta habitats, as well as the landscape processes and land uses that form and sustain those habitats. After describing the general approach, we describe a conceptual framework for understanding relationships among ecosystem processes, land uses, habitat conditions, and biota. This conceptual framework organizes the series of questions that must be answered, and identifies the purpose of each assessment method. We also describe the relationships among the different assessments, suggest a sequence in which the assessments can be conducted, and explain a series of questions that should be answered in the process of developing a habitat recovery plan.

Restoring Ecosystems to Support Recovery of Listed Salmon

Over the past decade, many scientists have pointed out that the listing of salmon and other species as threatened or endangered is largely a result of trying to manage individual species and habitat characteristics rather than managing whole ecosystems (e.g., Doppelt et al. 1993, Frissell et al. 1997). It has also been recognized by scientists and managers alike that restoration that carefully considers the watershed or ecosystem context is more likely to be successful at restoring individual or multiple species and preventing the demise of others

(Nehlsen et al. 1991, Doppelt et al. 1993, FEMAT 1993, Lichatowich et al. 1995, Reeves et al. 1995, Beechie et al. 1996, Moore 1997). These conclusions suggest that habitat recovery planning will require assessments of disruptions to ecosystem functions and biological integrity, which have reduced the productive capacity of Pacific Northwest river systems and are partly responsible for the declines in salmon abundance. The goal of such assessments is to identify alterations of key processes that affect stream ecosystems, and specify the management actions required to restore those processes that sustain aquatic habitats and support biological integrity (e.g., FEMAT 1993, Moore 1997, Quigley and Abelbide 1997, Beechie and Bolton 1999). In this approach, restoring specific salmon populations (or any other single organism) is subordinate to the goal of restoring the aquatic ecosystem that supports multiple salmon species. In addition, information on habitat changes or conditions that limit specific salmon populations can be useful for identifying actions that may have the greatest effect on salmon recovery (e.g., Reeves et al. 1991), or for helping to set population and ESU recovery goals. As long as all restoration actions are consistent with the overriding goal of restoring ecosystem processes and functions, habitats will be restored for multiple species, but in a sequence that favors one over the others.

For this report, the ecosystem approach to salmon recovery planning includes two main assessment elements: analysis of landscape and ecosystem factors to help set recovery goals, and analysis of disrupted ecosystem processes to identify watershed and aquatic habitat recovery actions. Each type of analysis relies on a conceptual framework describing general relationships among land uses, landscape characteristics, aquatic habitat, and biological responses (Figure 2-1). This framework illustrates that landscape processes and land uses alter aquatic habitats, which in turn alter aquatic communities or populations. Therefore, aquatic habitat conditions can be viewed as the link between landscapes and fish populations. Making these relationships explicit allows us to organize analyses of ecosystem processes and functions in a way that brings greater clarity of purpose to each analysis, as well as a better understanding of how the results of each analysis are to be used in recovery planning.

Scientific Basis for an Ecosystem Approach

The scientific basis for this approach can be summarized in two important characteristics of salmon and their habitats (Beechie et al. 1996):

1. Salmonid stocks are adapted to local environmental conditions, including the dynamic nature of their environment (Miller and Brannon 1982, Healey 1991, Reeves et al. 1995).
2. Spatial and temporal variations in landscape processes create a dynamic mosaic of habitat conditions in a river network (e.g., Naiman et al. 1992).

These statements imply that salmonid species or stocks are adapted to spatially and temporally variable habitats (Beechie et al. 1996), and may further imply that such environmental variability is important to the long-term survival of stocks or races (Reeves et al. 1995). Perhaps most importantly, different salmon populations (even some located very close to each other) are adapted to the different spatial and temporal sequences of habitat conditions found in each watershed, which influences life history diversity across an ESU.

Because salmonids are adapted to spatially and temporally varied habitat conditions, it does not make sense to manage for the same conditions in all locations, or to expect conditions to remain constant in any single location. This has been recognized in scientific critiques of many management issues in the past decade, including “one-size-fits-all” habitat standards (Bisson et al. 1997), not managing for spatial or temporal variation in habitats (Reeves et al. 1995, Bisson et al. 1997), and addressing symptoms of a disrupted ecosystem rather than the causes (Frissell and Nawa 1992, Spence et al. 1996). Those approaches generally do not consider that local populations are adapted to the natural potential habitat conditions within their range, and that those conditions vary in space and time. By contrast, identifying the root causes of degradation (i.e., disruptions to ecosystem processes and functions) focuses restoration on those processes that form and sustain habitats, which allows each part of the river network to express its natural potential habitat, and helps conserve and restore the natural spatial and temporal variation of habitats to which salmon are adapted.

We stress that identifying the root causes of ecosystem degradation is important for two main reasons. First, we do not understand most of the linkages between landscapes, habitat, and salmon populations with any great certainty, and we cannot predict exactly how land uses alter habitat conditions or how those habitat changes alter salmon populations. In fact, it can be argued that we are not yet even aware of all the aspects of aquatic ecosystems that significantly affect salmon populations. This lack of knowledge has in the past led to significant habitat degradation. For example, the role of wood debris in habitat formation was poorly understood until the 1970s. Consequently, removal of wood from rivers for navigation over the past 150 years has resulted in dramatic alteration of river habitats (Sedell and Luchessa 1982, Collins and Montgomery 2002), and as recently as the 1980s biologists recommended widespread wood removal to help adult salmon migrate upriver. While wood removal is far less common presently (but still occurs), the example serves to illustrate that we could have avoided significant habitat loss by choosing management actions that preserved riparian forest processes and natural wood functions in channels, even without understanding the value of wood in aquatic ecosystems.

Second, traditional restoration actions such as bank protection or spawning gravel placement attempt to build habitats that do not move in space or time, whereas natural habitats are often created by movement of river channels, wood debris, and sediment. Therefore, many restoration actions fail to restore habitats because they do recognize the integrated nature of physical and ecological processes in watersheds (Frissell and Nawa 1992, Beechie et al. 1996). This lack of knowledge leads to two main types of failure: 1) site-prescribed engineering solutions can be overwhelmed by altered watershed processes that are far removed from degraded habitats (e.g., increased sediment supply from upslope sources can bury engineered structures and pools), or 2) such measures can prevent habitat formation that would otherwise naturally occur (e.g., bank protection prevents formation of new off-channel habitats). Avoiding these types of project failure requires that we focus on restoring ecosystem processes and functions that form and sustain salmonid habitats, rather than on the habitats themselves.

Many organizations have recently adopted approaches to salmon habitat restoration that have a watershed or process-based approach (e.g., Moore 1997, SWC 1998, OWEB 1999a, JNRC 2001), which should help avoid some of the mistakes just described. However, many local groups continue to identify restoration projects in an opportunistic fashion, making difficult to assemble a broader understanding of habitat degradation and decline of listed species. Without this larger context, proposed projects are often disconnected from each other and fail to address the most important habitat losses. We encourage further development of holistic assessment approaches at the local level (e.g., SWC 1998) in order to more effectively utilize funding and resources allocated for salmon recovery.

Practical Considerations

There are a variety of elements in a salmon recovery plan that can be addressed through systematic watershed assessments. First, these assessments can contribute to the three tasks listed in the TRT guidance document in the following ways:

1) understanding fish and habitat relationships thorough correlation analyses can help in setting population recovery goals (Phase I planning),
2) assessments of historical and current habitat abundance and quality are important for identifying factors causing decline and limiting factors (Phase II planning), and
3) assessments of altered habitat-forming processes can help identify early actions for ecosystem recovery (also Phase II planning).

In combination, these assessments provide a comprehensive understanding of actions that are likely to improve population performance of listed salmon. From these analyses both regional and site-specific plans for ecosystem restoration can be developed.

Beyond the TRT guidance tasks, systematic watershed assessments provide a complete picture of disrupted habitat-forming processes as well as an inventory of degraded habitats. These assessments provide a watershed level understanding of restoration needs that can be used as a basis for judgments made in consultations by the agencies, for evaluating HCPs or programmatic actions that may receive incidental take permits, or for evaluating proposed habitat restoration projects (see also Golde in preparation). In addition, systematic watershed assessments can provide information that is useful for addressing management actions related to the Clean Water Act. Over the past two decades, scientists have proposed several new management approaches that seek to alleviate failures associated with managing individual habitats and species. Understanding and managing for biological integrity of running waters became an accepted approach for diagnosing and monitoring problems related to the Clean Water Act (CWA) (Karr 1991), and more holistic management approaches have been described as watershed management (e.g., Swanson 1981), ecosystem management (e.g., Johnson et al. 1985), and managing for biodiversity (e.g., McNeely et al. 1990).

Currently, the region is faced with widespread salmon listings under the ESA as well as numerous listings of impaired water bodies under the CWA. Understandably, most jurisdictions are seeking management approaches that will satisfy the requirements of both Acts in order to avoid duplication of efforts in the regulatory arena. With respect to legislation, an ecosystem approach will accomplish the habitat-related purposes of both the ESA and CWA, which are to conserve the ecosystems upon which listed species depend (ESA), and to restore and maintain the physical, chemical, and biological integrity of the nation's waters (CWA). Because watershed processes and biodiversity are essential components of the aquatic ecosystems that support salmon, managing an entire riverine ecosystem requires that watershed processes be restored and maintained, and that the biodiversity of salmon and other organisms be conserved.

Watershed assessment approaches recently adopted by a variety of salmon recovery groups in the Pacific Northwest (e.g., SWC 1998, OWEB 1999a, JNRC 2001) should help address the parallel goals of the ESA and CWA more efficiently. These watershed assessment processes support an ecosystem approach, or at least include certain process-based components of an ecosystem approach (e.g., FEMAT 1993, WDNR 1995). While some of these processes were not specifically designed to help develop Pacific salmon recovery plans, they provide data that are relevant to understanding disruptions to watershed or ecosystem processes. To the extent that these assessments answer specific questions relevant to ecosystem recovery planning, their results can be used within the context of the approach described in this report.

Conceptual Framework for Habitat Recovery Planning

Developing the habitat elements of a salmon recovery plan requires considerable information about how different land uses have altered landscape and ecological processes that form and sustain the habitats upon which salmon depend. In this section we describe two basic groups of questions that must be answered to develop a habitat recovery plan (Table 2-1), and explain how each describes specific components of watershed function. The first set of questions concentrates on how habitats have changed since pre-settlement times and how those habitat changes have affected salmon and other biota. These questions motivate historical reconstructions of habitat types and abundance, as well as assessments of relationships between habitat and salmon population characteristics. The second group of questions focuses on identifying disruptions to ecosystem function and the types of habitat restoration that are necessary for ecosystem recovery. These questions motivate assessments that identify where the biological integrity of ecosystems has been degraded and where specific ecosystem processes or functions are disrupted.

Key Assessments for Habitat Recovery Planning

For organizational purposes it is useful to diagram the relationships between the various assessments that can be used to inform recovery planning (Figure 2-1). The first set of questions (those regarding changes in habitat and salmon populations) also fall into two groups: 1) assessments that quantify habitat change and then use habitat-based models to estimate changes in fish populations (e.g., limiting factors analysis, life-cycle models, EDT), and 2) correlation analyses that relate landscape and land use characteristics to fish population performance without directly quantifying changes to habitat (e.g., SWAM). It should be noted that neither of these assessments directly identifies causes of habitat degradation or specific restoration actions. However, these assessments have several important uses in Phase I recovery planning, or setting the recovery goals for the ESU and each population within it. First, they provide habitat-based estimates of potential population size for comparison to estimates from population viability analyses (see McElhany et al. 2000 for background on use of population viability analyses in describing VSPs). Second, they provide insights into potential changes in life history diversity by identifying losses of important habitat types. And third, the ESU-wide correlation analyses can be used to help identify which populations are most constrained by habitat loss and therefore may be most difficult to recover.

The second set of questions (those regarding ecosystem functions and biological integrity) can be separated into two components: 1) diagnosing aquatic ecosystem impairment through a multi-metric biological indicator such as the benthic index of biological integrity (B-IBI, Karr 1991), and 2) diagnosing causes of ecosystem impairment through specific assessments of watershed and ecosystem function (Beechie and Bolton 1999). Assessing the biological integrity of aquatic ecosystems provides two important pieces of information required in a recovery plan. First it identifies where the aquatic ecosystem has been impaired, and second it can suggest which types of ecosystem functions may be impaired. A multi-metric index such as B-IBI assesses the biota directly, and results of these assessments can be correlated with landscape and land use factors to indicate potential causes of impaired biological integrity. Assessments that correlate landscape and land use characteristics with population attributes (e.g., SWAM) can indicate which habitat changes are most likely responsible for declines in salmon populations, and therefore which broad categories of restoration actions are most likely to result in increased salmon populations. Direct assessments of ecosystem processes that form salmon habitats identify causes of degradation, as well as restoration actions that are required to recover ecosystem functions and biological integrity.

Table 2-1. Primary questions to answer in developing habitat recovery plans.

Question	Analysis area	Data types
Phase I questions: Assessing changes in habitat availability and potential impacts on population characteristics		
How might habitat changes have altered the abundance of individual populations?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
How might habitat changes have altered the productivity of individual populations?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
How might habitat changes have altered the diversity of life history patterns?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
How might habitat changes have altered the spatial structure of populations?	ESU or Watershed	ESU: mainly remote sensing Watershed: mainly field
What scenarios of habitat characteristics would support a viable ESU (viable meaning adequate levels of all 4 VSP parameters)?	ESU	Mainly remote sensing
Phase II questions: Assessing disruptions to ecosystem functions and biological integrity		
Where has biological integrity been degraded?	Watershed	Field
Where have watershed processes and ecosystem functions been impaired?	Watershed	Field/remote sensing

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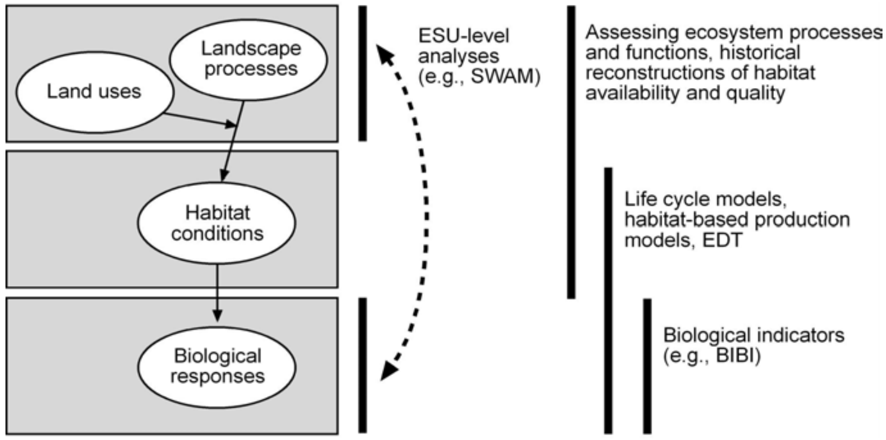

Figure 2-1. Schematic diagram of linkages among landscape processes, land use, habitat change, and biological responses (adapted from Beechie et al. In press). Assessing the biological response directly (e.g., using a biological indicator) identifies where ecosystem functions have been impaired, and may suggest causes of impairment. Assessments of habitat loss and resultant salmon population declines can be conducted by relating historical and current habitat abundance and condition to salmon utilization and survival. Assessing disrupted ecosystem functions and processes identifies causes of habitat change that result in diminished biological integrity and declines in salmon populations. For ESU-wide analyses of land use effects on salmon populations, landscape and land use factors can be correlated with indicators of population performance (e.g., SWAM-like analyses).

Ecoregional Differences in Assessments for Habitat Recovery Planning

The Pacific Northwest region encompasses a wide range of environments, which have been classified as a nested set of ecoregions with varying levels of detail (CEC 1997) (Figure 2-2). At the coarsest levels (Levels I and II), these ecoregions denote areas within which climate, lithology, topography, and ecosystems are generally similar (e.g., USEPA 2000, updated from Omernik 1987). Because climate, lithology, and topography are ultimate controls driving long-term habitat forming processes and stream habitats (Figure 2-3), ecoregions also provide a basis for identifying which ecosystem processes are locally more important and should be analyzed in ecosystem recovery planning. For example, an assessment technique that is suitable for identifying disturbances to wood recruitment and shade functions of dense riparian forests in coastal areas may need modification for understanding disturbances to the riparian functions of sparse hardwood forests or grasslands in the Columbia Plateau (e.g., FEMAT 1993, WDNR 1996).

For the Pacific Northwest and northern California, Level II ecoregions identify broad suites of landscape and climate characteristics for Marine Northwestern Coastal Forests, drier Northwestern Forested Mountains, and semi-arid to arid Western Deserts (Table 2-2) (CEC 1997). It should be recognized that that there is considerable local variation in climate and terrain within each Level II ecoregion, and that there exceptions to all classifications described here. Nevertheless, the major differences among these ecoregions are useful for illustrating how assessments may vary across the Pacific Northwest. Basic differences in ESU level and watershed level analyses for identifying habitat actions are shown in Table 2-3. In general, the same categories of assessments must be conducted regardless of ecoregion (e.g., sediment supply, riparian functions, isolated habitats), but the specific processes or mechanisms addressed may vary from one ecoregion to another. For example, sediment supply to the stream network should be evaluated in any watershed, but certain processes of sediment supply may be emphasized depending on location. Sediment supply is dominated by landsliding in wet mountain forests of the Coast Range and Cascade Mountains (e.g., Sidle et al. 1985), so understanding land use effects on landslide rates and sediment volumes is critical to necessary restoration actions such as road decommissioning or reconstruction. By contrast, sediment supply in dry rangelands of the Columbia plateau is more a function of surface erosion and gullying (e.g., Kaiser 1967, Peacock 1994), especially where soils are bare for some portion of the year due to agricultural practices. In these areas, assessing sediment impacts will focus more on changes in surface erosion rates and volumes in order to identify where modification of agricultural practices may reduce sediment supplies. These and other analyses of watershed processes will be described in more detail in the "Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions" and "Prioritizing Potential Restoration Actions Within Watersheds" sections of this document.

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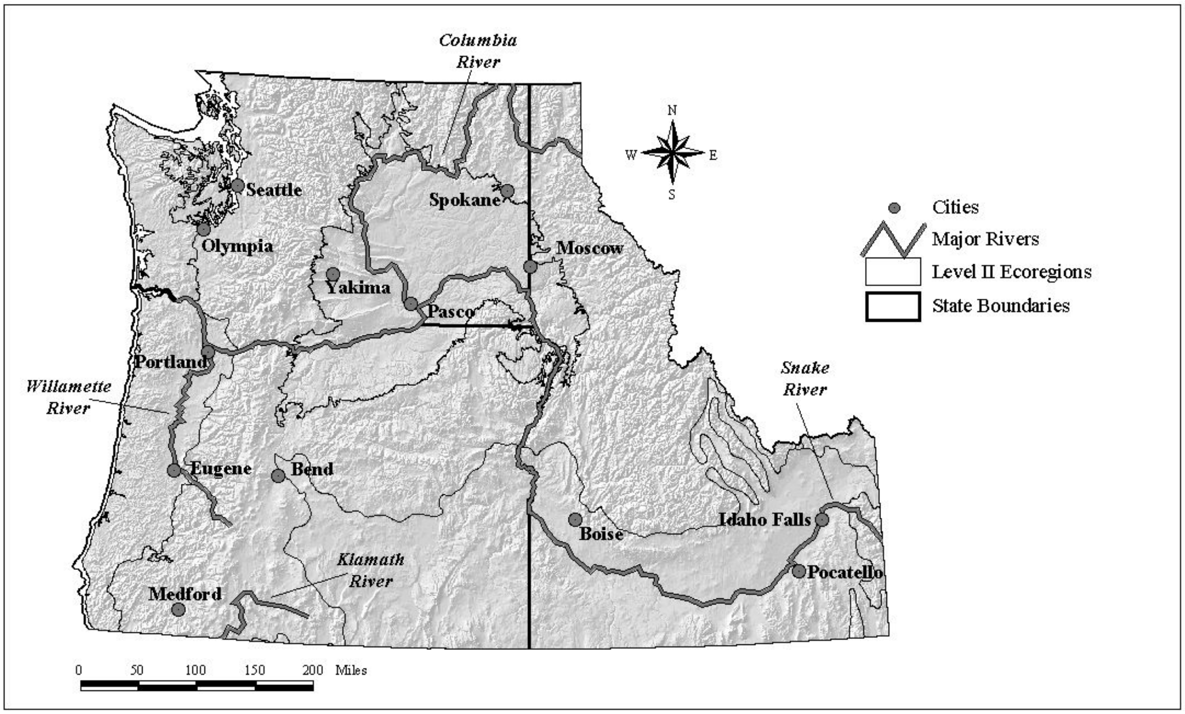

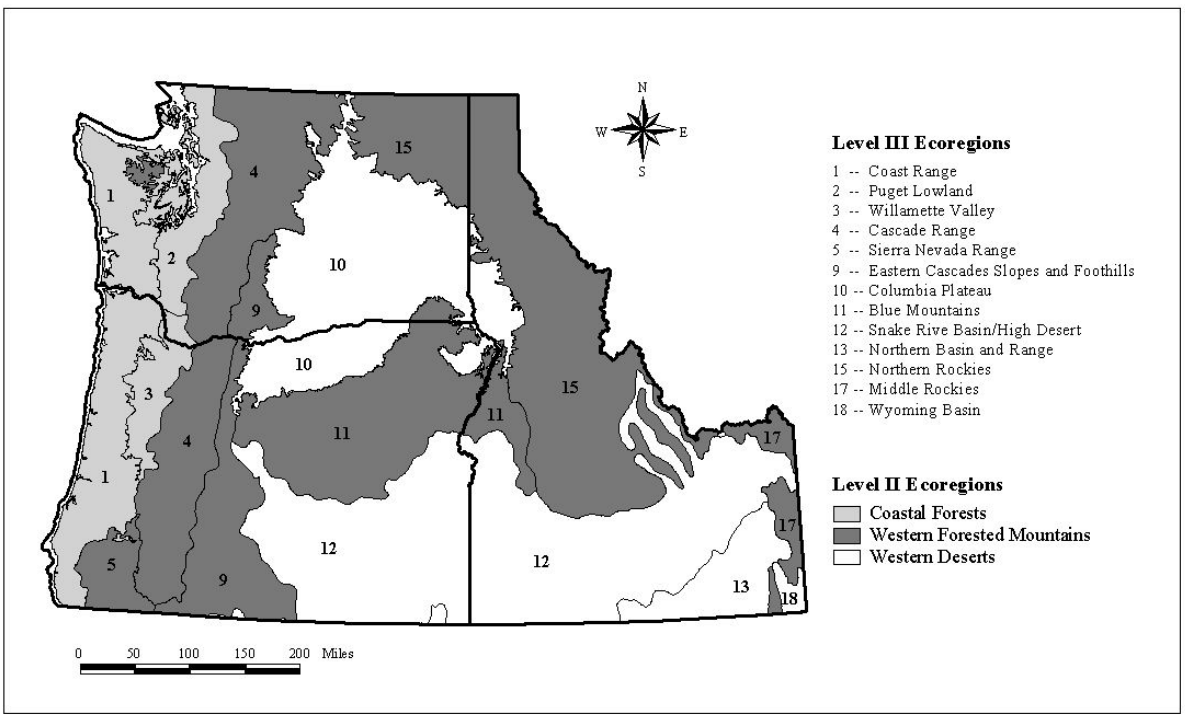

Figure 2-2. Level II and Level III ecoregions of the Pacific Northwest. See Table 2-2 for general descriptions.

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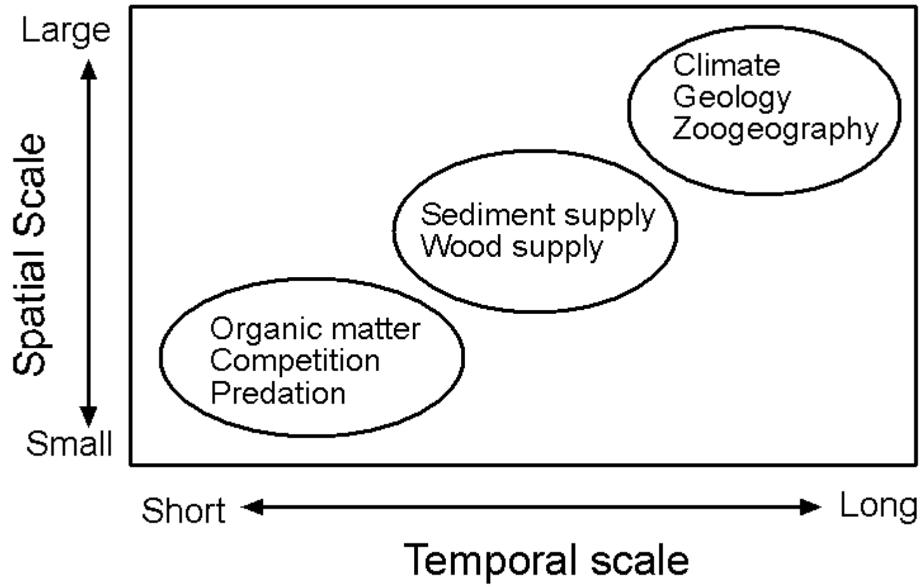

Figure 2-3. Spatial and temporal scales of factors that control habitat conditions and fish production in streams (adapted from Naiman et al. 1992).

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Table 2-2. Regional differences in landscape and climate characteristics of Level III ecoregions within the Pacific Northwest. Note that some ecoregions extend well outside the Pacific Northwest and exhibit a broader range of characteristics than described in this table.

Level II ecoregion	Level III ecoregion	General characteristics
Western Deserts (semi-arid to arid rangelands)	• Northern Basin and Range • Snake River Plain • Columbia Plateau	• Annual precipitation <50 cm • Snowmelt dominated flood regime • Gentle slopes • Grassland, sage • Sediment supply dominated by surface erosion, gullying • Low to moderate wood abundance in channels
Western Forested Mountains	• Eastern Cascade Slopes and Foothills • Blue Mountains • Middle Rockies • Idaho Batholith • Northern Rockies	• Annual precipitation 50 to 80 cm • Snowmelt dominated flood regime • Steep slopes, moderately dissected • Sparse conifer forests • Sediment supply dominated by surface erosion, gullying • Moderately abundant wood in channels
Coastal Forests	• Klamath Mountains • Coast Range • Cascades • North Cascades • Willamette Valley • Puget Lowland	• Annual Precipitation 75-400 cm • Rain dominated flood regime • Steep, heavily dissected terrain, except in flatter Willamette and Puget lowlands • Sediment supply dominated by mass wasting • Dense conifer forests • Abundant wood in channels

*****FOR REVIEW ONLY*****

Table 2-3. Regional differences in dominant ecosystem processes and assessments in the Pacific Northwest. This table is intended only to illustrate that different processes and assessments should be emphasized in different ecoregions. Important ecosystem processes vary within Level II ecoregions, and watershed-level assessments should target those processes that are locally important within each watershed. (Note that the Columbia River estuary is in the Coastal Forest Ecoregion, but also affects Columbia River stocks in the Western Deserts and Western Forested Mountains.)

	Level II ecoregion		
	Western Deserts	Western Forested Mountains	Coastal Forests
Sediment	Gullying and surface erosion (especially in agricultural areas)	Mass wasting and gullying	Mass wasting (surface erosion in agricultural lowlands)
Flood hydrology	Snowmelt dominated flood regime	Snowmelt dominated flood regime	Rain and rain-on-snow flood regime
Low flow hydrology	Diversions and dams common	Diversions common and dams	Diversions and dams less common
Riparian functions	Grasses and shrubs, some forest in floodplains	Sparse forests, shade a dominant function	Dense forests, wood recruitment a dominant function
Isolated habitat	Culverts, dams and dikes common	Culverts, dams and dikes common	Culverts, dams and dikes common
Estuary function	NA (Columbia estuary should be assessed in relation to freshwater habitats)	NA (Columbia estuary should be assessed in relation to freshwater habitats)	Severe impacts in agricultural and urban areas
Biological integrity	Especially important in urban and agricultural areas	Especially important in urban and agricultural areas	Especially important in urban and agricultural areas

Sequencing the Assessments

Figure 2-1 illustrates that natural landscape processes are linked to biological responses by their effects on aquatic habitats, and that land uses alter habitat conditions by disrupting natural processes. However, it does not identify the order which analyses should be conducted in recovery planning. In general, Phase I assessments should be conducted as quickly as possible in order to be of use in setting the population recovery goals. ESU-wide assessments of correlations among landscape attributes and fish population characteristics (e.g., abundance, life history patterns) and watershed-level assessments of habitat availability for different life history stages can be conducted simultaneously. The former provides information that is useful in setting goals for recovery the entire ESU, whereas the latter is useful in recovery goals for individual populations. Ultimately, this information may also be useful in prioritizing restorations actions that are identified in Phase II recovery planning.

Sequencing of the Phase II assessments is determined by management needs and the rate at which necessary information can be acquired. Habitat restoration is currently underway, and the management need for reliable identification of important habitat recovery actions is immediate. However, identification of site-specific restoration actions requires time-consuming field inventories that are far from complete in many cases. Given this immediate need and the relatively long time frame required to complete the many required inventories, the first assessments should be rapid assessments of land uses, habitat change, and biological responses across entire ESUs (Table 2-2). These assessments (discussed in detail in "Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions" section) often rely on remote sensing data such as satellite imagery or digital elevation models as primary data sources (e.g., Lunetta et al. 1997). Other coarse resolution geospatial data sets (such as maps of land use types, mean annual temperature, or mean annual precipitation) are also frequently used. Site-specific data such as barrier inventories or stream habitat data may also be available in some areas, but in most cases these data exist for relatively few sites and must be extrapolated to most of the ESU. Therefore, ESU-level questions typically will be addressed with broad correlative studies that relate landscape and land use attributes to fish population parameters, without sit-specific understanding of habitat changes that have occurred in streams. Referring to Figure 2-1, these assessments will relate landscape attributes and land use practices directly to fish abundance or survival, and ignore causal mechanisms that link landscape processes and land uses to habitat change or habitat change to fish population response. These ESU-wide assessments can indicate general patterns of population declines resulting from different land uses, but the data are generally too coarse to allow detailed analyses of habitat change and its effects on fish populations (e.g., Lunetta et al. 1997, Pess et al. 1999a).

Watershed-level assessments are more detailed and time-consuming, and managers should expect such that such inventories and assessments will take several years to complete. Such efforts should focus on the full range of potential habitats in the planning area, including freshwater, estuary, and nearshore marine areas. The conceptual model itself does not limit the spatial or temporal resolution of the results. Rather, specificity of results is driven by the resolution of the data, which in turn is driven by the assessment goals. For example, coarse resolution remote sensing data might be used to investigate patterns in broad patterns of riparian alteration within a watershed, which can be used to estimate the overall magnitude of the problem and to estimate restoration costs for a watershed (e.g., the riparian assessments in Appendix B). However, these data and assessments do not have sufficient detail to pinpoint the exact nature of the problem at any single site, and field inventories must be conducted in order to identify site-specific recovery actions. Similarly, other types of restoration actions such as removing migration barriers or landslide hazard reduction require field inventories to identify specific actions. (Note also that this remains true even if EDT analyses have been completed.) As a general rule, larger scale assessments help managers understand where landscape processes have been most altered, and site-specific inventories identify specific actions that are needed to restore those processes.

Implementing Recovery Actions

We support habitat recovery strategies that have an overarching goal consistent with the ESA (i.e., conserve the ecosystems upon which listed depend), as well as the CWA (i.e., protect the biological integrity of aquatic systems). Consistency with the purposes of these two acts will simplify the assessments needed to identify necessary habitat protection and restoration actions, and help avoid conflicts that arise from managing for the specific (and often conflicting) habitat requirements of multiple species. Examples of such a goal are to "protect and restore the processes that form and sustain habitats to which salmonid stocks are adapted" (Skagit Watershed Council 2001), "restoration and protection of habitat conditions and processes upon which the fish depend" (Lower Columbia Fish Recovery Board 2001), or "to have a diversity of habitats and natural processes necessary to sustain healthy populations of native species" (Willamette Restoration Strategy 2001).

Strategies should also have clearly stated near-term and long-term objectives. Near-term objectives typically should include taking those actions that we already know are consistent with conservation of ecosystems that support salmon, such as inventory and removal of migration blockages, habitat protection through easements or purchase, and protection and restoration of riparian forests and their functions (see also the "Prioritizing Potential Restoration Actions Within Watersheds" section for more detail). Longer term objectives may include targeted assessments or management experiments to clarify which actions are most beneficial to aquatic ecosystems, as well as implementing larger restoration projects that require changes in infrastructure or land uses, such as modifying levee systems to reopen access to estuary habitats and tidal channels.

Once the goals and objectives of the strategy are defined, it is important to identify specific questions that must be answered in order to proceed with protecting and restoring aquatic ecosystems. These questions will generally be more targeted and specific than those listed in Table 2-1. For example, several of the questions listed in Table 2-1 require that we know how much habitat has been blocked to salmon access. An obvious set of questions can be written to address this topic, including 1) where are the blockages to salmon migration, 2) how much habitat is above each blockage, and 3) how much will it cost to repair each blockage? These questions drive the need for a comprehensive and systematic inventory of stream crossing structures (including small dams, levees, tide gates, culverts, etc.), which can be conducted using widely accepted standard methods such as the barrier inventory methodology of Washington Department of Fish and Wildlife (WDFW 1998). Similar sets of questions can be written for other aspects of ecosystems such as riparian functions or changes in supply of sediment, as well as for understanding the condition of aquatic habitats through inventories of habitat and biological indicators such as invertebrate or fish communities.

We recognize that many jurisdictions and restoration groups are overwhelmed by the many assessments and inventories needed to enact a restoration strategy. However, the task is not so daunting as it first appears, provided that one recognizes the long-term nature of it. In essence, the steps are:

1. Develop the restoration strategy.
2. Identify what you need to know to implement the strategy.
3. Identify what you already know.
4. Act on what you know while you conduct inventories that fill in the data gaps.
5. Revise the plan and actions as new information comes in.

Steps 1 and 2 describe the overall restoration strategy and the types of information required to implement it. In some respects this is the most difficult step. In this section we explained a restoration strategy that focuses on restoring the ecosystems upon which salmon depend, and briefly described the questions that must be answered in order to restore them. This general approach is supported by the scientific literature, and alleviates several management conflicts that may arise from attempts to separately manage individual species and habitat characteristics. Adopting a conceptual framework similar to that described here will help TRTs and local jurisdictions to more rapidly move through strategy development, organize their information needs, and ultimately better understand the habitat changes that have contributed to salmon declines in the region.

Updating the Recovery Plan

[In the "Updating the Recovery Plan" **subsection, to be further developed**, we discuss interim actions that are known to be consistent with protecting and restoring the ecosystems upon which salmon depend, and also describe how to use the information collected in various assessments to prioritize appropriate restoration actions. This requires examining all the questions identified from the strategy and listing which questions are already answered or partly answered. Subsequently, local jurisdictions can identify areas where causes of habitat degradation or loss are already known, and therefore where habitat restoration or protection actions can be implemented immediately. Over the long term, identification of habitat protection and restoration actions relies on systematic inventories of causes of biological conditions in streams and causes of habitat loss. These inventories must be comprehensive and systematic in order to reliably identify the suite of possible restoration actions and to evaluate their relative importance to ecosystem recovery. (See also the "Prioritizing Potential Restoration Actions Within Watersheds" section and Appendix B).]

HABITAT ANALYSES FOR PHASE I RECOVERY PLANNING: SETTING RECOVERY GOALS

Beth L. Sanderson, E. Ashley Steel, Tim Beechie, George Pess, Mindi Sheer

Recovery goals consist of a combination of quantitative and qualitative targets for fish populations, their habitats, and management outcomes (McElhany et al. 2000). Goals might include numerical fish population targets, numerical population trend targets, qualitative targets for spatial distribution of populations, and quantitative and qualitative population diversity, habitat quality, or management outcome targets (McElhany et al. 2000). In this section, we consider how habitat analyses can inform the development of quantitative fish population viability goals for salmon populations, watersheds, and ESUs. ESU-level analyses are used to examine the quantity and quality of habitat across numerous populations within ESUs, whereas watershed-level analyses focus on questions that are specific to individual populations. These large- and small-scale studies differ in the kinds of information, data, and approaches used to contrast the ability of current habitats to support productive fish populations to that of historic habitats.

As ESU and population viability goals are set, we must simultaneously evaluate whether current, historic, or potential habitat might be sufficient to support populations of the desired size, as well as whether the distribution of available habitats can support the desired diversity and spatial structure of salmon populations. For any of these analyses it is critical to select habitat measures that (1) can be linked to population performance and (2) are sensitive to land use changes or restoration actions. Habitat measures (physical, chemical, and/or biological) that meet these criteria facilitate an understanding of how land uses or restoration actions change habitats, and how those habitat changes in turn create population responses in salmon (see also Figure 2-1). This is true whether the analysis is conducted with coarse resolution data covering entire ESUs, or finer resolution data available only in certain watersheds.

Because habitats and biota are hierarchically structured (see Figures 2-3 and 6-1), it is useful view habitat data in the context of a hierarchical classification system such as that illustrated in Table 3-1. With such a classification of habitats, results of coarse-resolution analyses across entire ESUs can be linked to fine-resolution analyses within individual watersheds. Moreover, this hierarchical structure allows one to construct simple predictive models for estimating abundance and distribution of fine-resolution habitats based on coarse-resolution data (e.g., Lunetta et al. 1997).

ESU-level Analyses

ESU-analyses are different from watershed analyses in that they ask questions and analyze data that span the area of entire ESUs. The area of existing ESUs ranges from 4500-24000 square miles. The number of independent salmon populations within each ESU varies from as few as one to more than 30 (see http://www.nwfsc.noaa.gov/cbd/trt/). Because these analyses encompass large geographic areas and often more than one salmon population, the goals and data needs are inherently different from efforts directed at watershed, stream or reach scales.

There are a number of reasons for conducting ESU-scale analyses. First, from a biological point of view, salmon recovery must occur at the ESU scale. For this to happen, we need to answer questions, formulate hypotheses, and develop recovery scenarios for the entire ESU. At the moment, fine-resolution data (e.g., habitat typing, barrier inventories, etc.) are not comprehensively available across the large geographic areas these analyses must cover. However, recovery planning efforts need answers before new field data can be collected. These ESU-scale analyses can use currently available coarser scale data to address questions that span large geographic areas in a relatively short period of time (see Tables 3-2 and 3-3). Second, standardized and consistent approaches are critical for comparisons both within and across ESUs. And last, ESU-scale analyses act as a catalyst for further research and analysis by organizing existing data, identifying broad patterns and generating new hypotheses.

ESU-scale analyses are needed to provide initial answers to the Level I recovery planning questions identified in the "An assessment Approach for Habitat Recovery Planning" section and discussed previously in this section. Specifically, these analyses help answer the questions 1) How might habitat changes have altered the abundance, productivity, spatial structure and diversity of individual populations and ESUs? and 2) What scenarios of habitat characteristics would support a viable ESU? The nature of these questions, combined with the geographic scale of analyses, together means that ESU analyses, by nature, link landscape characteristics (land use and land form) to characteristics of the habitat and fish (Figure 2-1). Although specific approaches vary, ESU analyses generally aim to characterize large-scale patterns and identify correlations between landscape, habitat and fish characteristics. Furthermore, many ESU analyses aim to coarsely compare current and historical conditions.

Analyses for Abundance and Productivity Goals

How might habitat changes have altered the abundance, productivity, spatial structure and diversity of individual populations and ESUs? A variety of approaches have been used to examine relationships between salmon abundance/productivity and habitat characteristics at scales of the ESU or larger (Table 3-2). Many of these are correlation studies that link patterns of land cover and land use to either measures of fish abundance or survival, or measures of in-stream habitat quality. Fewer studies have tried to determine how changes in habitat may have altered the diversity and spatial structure of salmon populations and ESUs. In this section we describe two approaches to assessing how land uses might be related to salmon abundance. The first uses simple correlations among landscape/land use variables and salmon abundance to evaluate the relative quality of different stream reaches within a large study area (e.g., Pess et al. 2002, Feist et al. in prep). The second uses coarse resolution data to estimate historical and current habitat abundance within an ESU.

Correlation analyses

Examining available data on fish populations and habitat conditions is a first step toward understanding the ESU-level relationships between salmonid populations and the physical, chemical, and biological components of their habitat. Many potential metrics of population performance exist: genetic diversity, juvenile abundance, adult abundance, productivity, juvenile productivity per adult spawner, etc. Similarly, there are numerous habitat metrics: percent pools, watershed condition, water temperature, number or concentration of contaminants, etc. Correlation between population performance and habitat condition cannot identify cause and effect relationships because of correlation among habitat descriptors, correlation between landform and land use, and the potential for unmeasured variables to explain existing patterns. However, correlative analyses can be used to make predictions about where habitat conditions might limit or enhance salmon populations, to generate hypotheses for further testing, and to suggest important factors to control when setting up small-scale experiments, monitoring projects, or large management experiments (Figure 3-1).

One example of this type of analysis is the Salmonid Watershed Analysis Model (SWAM) (Pess et al. 2002, Feist et al. in revision). SWAM is a series of spatial and statistical analyses that relate salmonid population performance metrics in a particular basin to landscape and land use characteristics derived from existing geospatial data layers. This analysis identifies descriptors of landform and land use that are correlated with fish population performance in a given watershed. SWAM has been used in the Salmon River basin in Idaho (Feist et al. in review), the Snohomish River basin in Washington (Pess et al. 2002), and currently being used for analyses of the Willamette River basin in Oregon. In these three basins, SWAM used indices of adult fish abundance (redd counts and adult fish counts at index sites) as the metric of fish population performance, and multiple descriptors of landscape conditions across the entire watershed draining to the index reach as a surrogate for habitat condition. An alternate habitat metric, conditions in the riparian area directly associated with the index reach, was also tested in the Salmon and Snohomish basins.

The spatial and statistical analyses involved in the SWAM approach are comprised of six steps. First, conceptual mechanistic relationships between landscape features and population abundance during all freshwater life history stages are identified from the literature and from local habitat biologists. These conceptual relationships define the habitat data layers to be used as potential predictor variables. Second, spatial heterogeneity in the salmonid population data is examined to determine if certain areas in the basin consistently exhibit better population performance than other areas. Third, the landscape is characterized over the relevant area for each index reach. Fourth, landscape data layers are overlaid with the geo-referenced fish abundance (e.g., redd counts) data. Fifth, a statistical model is used to describe annually consistent relationships between landscape characteristics and fish abundance. And finally, these relationships are rolled into a predictive model and applied to the entire basin of interest (Figure 3-1). Many variations on the population and landscape metrics used in this type of analysis are possible. The best choice for a particular basin will be driven by available data.

SWAM or SWAM-like analyses provide a broad-brush estimation of fish occupancy within a basin and a first-cut estimate of the coarse-scale factors affecting abundance. In many cases, correlative models relating fish population performance to habitat conditions may also help identify the best remaining reaches or subwatersheds in a particular basin. If clear relationships between fish populations and habitat conditions exist, these analyses may suggest indicator habitat features and may identify areas that were historically productive. Ecological insights developed from these analyses may suggest likely habitat factors limiting population performance in a particular basin. Experience from these studies can be used to identify habitat characteristics to control when setting up experiments and M&E programs. Predictions of areas likely to support strong populations can suggest areas where detailed watershed assessments and habitat inventories should be conducted (see "Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions" section in the document).

Current and historical potential habitat

A second approach for evaluating the ability of multiple watersheds to support viable salmon populations is to examine the distribution and quantity of current and historic habitat across large geographic areas (e.g., Lunetta et al. 1997). This approach evaluates associations between fish and specific types of habitat for multiple watersheds and populations (e.g., comparing 20 demographic populations). These analyses use limited habitat survey or historical distribution information; ancillary topographic and hydrologic spatial data are used to supplement empirical reach-level data, and ultimately provide a tool for estimating habitat-based fish densities for multiple watersheds. In addition, these analyses rely primarily on geospatial data sets such as digital terrain models, Landsat imagery, and regional hydrography to predict physical stream features. The widespread availability of corresponding spatial datasets (remotely sensed imagery, GIS data and spatially-explicit modeling methods) permits the derivation of representative reach-level geomorphic and hydrologic information for multiple watersheds (Table 3-3). This information can be used in conjunction with field-based mapping data, such as anthropogenic modifications (e.g., Bortleson 1980), to refine estimates of currently and historically available habitat for different species. Subsequently, methods to refine historic habitat estimates can be used to estimate possible levels of historical fish densities across the watershed.

One of the major habitat alterations influencing anadromous salmonids in the Pacific Northwest is the modification of migration corridors with the placement of physical barriers and blockage of upstream or downstream migration. Anthropogenic barriers include dams, diversions and culverts. Barriers that impact stream flow and fish passage are often available as spatial databases, especially in larger streams, and in those streams historically occupied by ESA –listed salmonids. These can be useful in estimating natural and anthropogenically-induced upper limits to fish distribution, and ultimately, information on the current and historic habitat.

We have used a GIS based approach to quantify how barriers have changed current and historic habitat availability in the Willamette – Lower Columbia (WLC) Recovery Domain (drainage area of approximately 50,000 km^2). The analysis addressed three questions for all

ESA-listed ESUs and associated populations. 1) How many miles of stream habitat are unavailable or unreachable to fish due to the construction of anthropogenic barriers? 2) What proportion of pre- and post-barrier habitats would have been or are currently useable habitats for juvenile and adult salmonids? 3) Is current habitat sufficient to support viable salmonid populations? What fish densities would need to be maintained if only current, or current and historic habitat were available to support populations? Which areas could benefit the greatest from barrier-related recovery activities?

To answer these questions, field data (already in GIS format) were incorporated, and augmented with habitat-related data generated from computer models. Locations of in-stream barriers to fish migration, limited habitat survey data, fish distribution maps (based on field data and expert opinion), hydrologically correct streams, fish preference criteria (expert opinion), and variables derived from topographic landscape models were used to delineate currently inaccessible streams, and sections likely to be suitable for spawning and rearing for species of interest, and estimates were compared to independently completed population viability analyses (see Steel and Sheer [2002 draft???] for additional details).

In the Willamette/Lower Columbia ESU, a total of 2,600 barriers, fine-scale stream hydrologic GIS data, and current fish distribution maps were used to summarize the total number of inaccessible stream miles for all ESUs and populations within the Willamette/Lower Columbia (Figure 3-2). Overall, these individual barriers within the Willamette/Lower Columbia block between 1 and 2,000 km of stream habitat for individual populations. Currently inaccessible possible habitat (by population) ranges from 0–98% of the amount that was historically available to salmon. This barrier identification step provides a glimpse of the amount and type of habitats that have been lost due to barriers, but it does not address whether current or historically reachable stream habitats are useable or likely to be preferred by juvenile or adult fish. More precision and accuracy are needed to address issues of fish occupancy or densities within these areas.

This next step examines whether stream habitats are useful to fish by linking reach-level habitat features and the anthropogenic barrier data. We obtained reach-level habitat characteristic information from a procedure developed by Miller (2000) that uses a digital elevation model to generate stream-related hydrologic information. Variables available for all stream reaches included stream gradient, drainage area, reach length, valley floor width, side-slope gradient, and mean annual precipitation. Channel width was modeled using empirical observations and derived physical information using methods developed by Hyatt (unpubl. data). Channel gradient was identified as the best indicators of useable habitat. This useable habitat includes both prime and possible habitats, where prime is likely the most useable habitat. Additional criteria such as stream width can help generate more realistic estimates of prime and possible habitat total lengths. A series of channel gradient thresholds describing useable spawning and rearing habitat were created for each species. Stream reaches meeting specific gradient range thresholds were identified as possible or prime spawning and rearing habitat for winter steelhead, summer steelhead, chinook, and chum salmon (e.g., Figure 3-3). The total amount of useable habitat was summed for both historic and current stream reaches. For fall Chinook ESU, an average of 15% (0–98%, n = 21 populations) of prime habitat was located upstream of barriers. The average percent of prime habitat currently accessible to fall Chinook ranged from 2–92% (average = 59%, n = 21 populations).

The remaining question is whether the amount of useable habitat is sufficient to support viable salmon populations across the ESU. In the Puget Sound and WLC ESUs, demographic models were used to estimate target numbers for individual populations (see http://www.nwfsc.noaa.gov/cbd/trt/). The amount of useable habitat calculated as described above was used to verify if target numbers were realistic given the actual proportion of stream miles able and likely to be used by salmon, either currently or historically. Fish densities (calculated by dividing target numbers by total number of prime or possible stream miles) were divided into three categories, 1) clearly historically achievable, 2) clearly historically unachievable, and 3) historical achievability unknown. If the demographic modeling targets fall into category 1, we will accept the model-derived target as the necessary required criteria for recovery. If the target falls into category 2, we may need to look more carefully at the demographic modeling assumptions. If the target falls into category 3, further analysis will be required to evaluate the historical abundance of the population.

Analyses for Spatial Structure and Diversity Goals

As mentioned earlier, there have been few analyses of spatial structure and diversity of salmonids within ESUs (Table 3-2). However, two studies provide good examples of how one might approach the problem: the Interior Columbia Basin Ecosystem Management Project (ICBEMP 1997) and a characterization of the diversity of salmon in the Pacific Northwest (Waples et al. 2001). In a broad assessment of the Columbia River Basin, ICBEMP characterized the distribution of individual fish species, the patterns in fish species diversity and status, and their relationships to landscape characteristics. Their analyses indicate major declines in the number and distribution of salmonid species located in Columbia Basin subwatersheds. Anadromous fish species have been largely extirpated from large portions of their range; the consequences of this for ESU viability need to be addressed. Large-scale analyses such as

ICBEMP provide valuable information and insights into how spatial structure and diversity of salmon and other species have been altered from historic conditions.

Waples et al. (2001) characterized patterns of intraspecific diversity for Pacific Northwest salmon species along three major axes: ecology, life history and biochemical genetics. The ecology axis included characteristics of freshwater habitats that are of known importance to salmon (i.e., hydrography, temperature, vegetation, geology, etc.). The results indicated that the amount of diversity within a species was significantly related to the extent of ecological diversity experienced by that species. For example, diversity measures were lower for pink and chum salmon, which are limited in distribution to areas directly affected by Pleistocene glaciation. In contrast, chinook salmon and steelhead were broadly distributed and exhibited the greatest degree of diversity.

Watershed–level Analyses

For Level I recovery planning, watershed-level analyses are designed primarily to assist in setting abundance goals for recovery, although they can also shed light on the remaining three types of recovery goals (productivity, diversity, and spatial structure). Habitat analyses at the watershed-level are conducted using one of two general approaches: (1) a complex expert system model such as EDT, or (2) simpler empirical models relating landscape variables, habitat condition, and population attributes. In general terms, both approaches assess historic and current habitat conditions in a watershed, and use those assessments to estimate changes in probable salmon abundance at different life stages. However, the two approaches differ in that EDT relies on expert judgments of historic and current habitat conditions for a wide variety of rarely measured parameters, whereas the empirical models focus on very few parameters that can be measured or estimated by repeatable empirical relationships. In this section we briefly review these two approaches in the context of setting of abundance goals for recovery, and then describe how the same methods might contribute to setting goals for productivity, diversity, and spatial structure.

Analyses for Abundance or Productivity Goals

Ecosystem diagnosis and treatment

Ecosystem Diagnostic Treatment (EDT) is a complex salmon production model that has been used throughout the Pacific Northwest for salmon recovery planning. The conceptual basis for EDT is described in Lichatowich et al. (1995) and Mobrand et al. (1997), and the details of the model structure and its use are found in Lestelle et al. (1996). EDT is a habitat-based model that relies on expert opinion (best professional judgment) as environmental input data. It organizes environmental inputs, estimates habitat condition, and predicts fish population performance based on life-stage specific stock-recruit functions. Fish population performance is characterized using a "survival landscape," which is created by multiplying the life-stage functions together. The survival landscape is typically created for environmental states corresponding to two points in time – current and "historic," but may also be created for other states. The spatial unit of analysis is a reach (e.g., one river mile to several river miles).

There are three levels of information in an EDT analysis. Level 1 is the information that goes into EDT, which includes 45 environmental variables (or some subset thereof). The environmental variables that make up Level 1 inputs include actual data (e.g., data measured in the field), derived data (e.g., estimated from coarse-scale data), and anecdotal information (e.g., best professional judgment). Level 2 is the set of the biological rules that drive the model. Biological rules are working hypotheses about how a salmonid species at a given life stage will respond to a specific environmental variable. The shape of each biological rule is an exponential function. The sum of the functions results in a combined estimate of the survival of fish throughout their life cycle. The rules are based upon 1) literature review and 2) expert judgment. Expert judgment can come from consultations with nationally known or local "experts." The level of proof behind each rule is ranked based on the information underlying it.

Level 3 is the translation of the rules into 17 different biological performance attributes (variables) that give relative survival or performance by life stage. Each attribute is used to modify relative survival compared to an idealized or optimum condition that is the "historic template." Survival and performance by life stage are the sum of the 17 biological attributes, and are always a fraction that ranges between 0 and 1. The difference between the optimal and current is the relative survival landscape. The 0-1 relative survival parameter is then used to modify the productivity parameter of a Beverton-Holt recruitment function associated with each reach for each life stage. The modification is relative to the idealized historic condition.

The quantity of habitat required to support optimal and current capacity is also estimated. The optimal and current capacities by life stage for every reach (e.g., several kilometers of stream) are calculated by associating a "habitat type" with each reach, then multiplied by a "weight" equivalent to population density. These estimates (optimal and current) are used as the capacity parameter of the Beverton-Holt recruitment functions at each reach and each life stage. Beaverton-Holt recruitment functions (optimal and current) are generated for the entire system by summing the individual life stage recruitment functions. "Life-history diversity" is incorporated by specifying different "pathways" that lead to the reach and life-stage specific Beverton-Holt functions.

EDT evaluates potential management options by proposing changes in habitat attributes, and modeling the resultant changes to the productivity and capacity of a population. Thus, EDT can help organize existing expert opinion and limited data to create hypotheses about linkages among certain habitat conditions and fish productivity. However, bearing in mind that the underlying data and functional relationships are largely untested, it is important to recognize that confidence in the accuracy of any EDT outcome is low. As noted in Mobrand et al. (1997), this performance measure is an indicator of how favorable the environment is (or might become) for salmon to persist and abound, not a predictor of how many will return and when. Such predictors are unreliable, and consequently, performance measures based on short-term abundance responses are poor guides to decision making.

Estimating current and historical potential fish production

Watershed assessments for estimating current and historical habitat availability and production potential require associating seasonal fish use and survival with a habitat type, as well as quantifying areas of different habitat types (Reeves et al. 1989, Beechie et al. 1994). Quantifying fish use of different habitat types is based on studies designed to identify these relationships (e.g., Bisson et al. 1988), but it may not be necessary to develop new relationships between fish use and habitat types in each watershed if it can be demonstrated that fish use is similar to that of previous studies (Beechie et al. in press). For example, different juvenile Pacific salmon species spatially segregate into different habitats in a watershed, as illustrated in Figure 3-4 (Pess et al. in press). Steelhead (*Oncorhynchus mykiss*), coho (*O. kisutch*), and ocean-type chinook (*O. tshawytscha*) have higher densities in specific habitat types for the same life stages. Juvenile steelhead utilize all habitat types in roughly equal numbers, while juvenile coho have greater preference towards slower water habitat such as side channels and ponds, as evidenced by higher densities in those areas. Ocean-type juvenile chinook prefer mainstem and estuary environments. Similarly, adult Pacific salmon are also temporally and spatially segregated during the spawning life stage (Groot and Margolis 1991, Lichatowich 1999, Montgomery et al. 1999).

Unlike associations among fish densities and habitat types, habitat data are not transferable across watersheds because the natural potential of stream networks and land use effects on habitat condition vary by watershed (Frissell et al. 1986, Lunetta et al. 1997, Beechie et al. 2001, Collins and Montgomery 2001). Therefore, habitat inventories must be conducted separately in each river basin or planning area. Assessments for estimating historical and current habitat are conducted in three steps: 1) identify habitat types, 2) estimate historical and current habitat abundance by type, and 3) estimate production potential based on habitat-fish relationships.

A habitat classification system suitable for estimating historical and current habitat and potential fish production must have two main attributes. First, analysts must be able to associate fish abundance and survival with each habitat type in order to estimate total fish abundance in a watershed. Second, it must be possible to quantify historical and current habitat areas in order to estimate changes in potential production over time. We recommend a suite of habitat types at two hierarchical scales, similar to that shown in Table 3-1. The coarser resolution of habitat types can be mapped from remotely sensed data at the reach scale (e.g., topographic maps, aerial photography, or satellite information), whereas the finer resolution of habitat types must be identified in the field at the habitat-unit scale (sometimes with the aid of aerial photography). Because these typing systems are nested, all reaches within a watershed can be stratified by landscape and land use factors using the remotely sensed coarse-resolution data, and reaches within each strata can be subsampled to develop an understanding of habitat types within each reach type. This hierarchical relationship enables extrapolation of habitat conditions for unsampled reaches within the watershed. Stratification of reach types may include several different landscape and land use factors, although a relatively small number of strata are desirable to reduce the complexity and number of assumptions and calculations. For example, tributary reaches may be stratified simply by slope and land use in order to identify changes in pool area as a result of land uses (Beechie et al. 2001). However, the same slope classes are not particularly relevant for large rivers, where some combination of slope and discharge may be more useful in predicting natural channel patterns (e.g., Leopold et al. 1964).

Methods for estimating current and historical habitat abundance differ among habitat types. Therefore, it is not possible to describe a single methodology for assessing changes for all reach types. Instead, we provide an overview of different approaches that one might use for assessing habitat conditions historically and at present (Table 3-4), along with references that provide greater detail on specific methods.

Once the habitat changes have been quantified, changes in potential population sizes can be estimated for specific life stages. Here we focus on juvenile production as an example, although similar approaches are useful for other life stages as well. As described in Reeves in et al. (1989), smolt production potential from a given habitat type or area is calculated as:

$$\text{habitat area} \times \text{average fish density} \times \text{survival to smolt.} \qquad (1)$$

However, comparing the impact of different habitat alterations on smolt production potential requires making separate estimates for each habitat type. Thus, the production potential of a habitat for each life stage (e.g., spawning, egg to fry, summer rearing, winter rearing, smolt migration) can be expressed mathematically as:

$$N = \left(\sum_{i=1}^{k} \left(\left(\sum_{j=1}^{n} A_{ij} \right) \times d_i \right) \right) \tag{2}$$

where ΣA_{ij} is the sum of areas of all habitat units (j =1 through n) of type i, and d_i is the density of fish in habitat type i (Beechie et al. in press). To compare capacities among life stages and identify which habitats may be limiting smolt production, the population estimate (N) for each life stage in a given habitat is multiplied by density independent survival to smolt stage so the capacities can be compared in terms of number of smolts ultimately produced (Reeves et al. 1989). Equation 2 can also be used to estimate historical spawner capacity (or potential population size at other life stages) based on estimates of historical habitat availability (Appendix A). Both spawning and rearing capacities can then be incorporated into assessments of factors that limit population size.

Analyses for Spatial Structure and Diversity Goals

Examples of watershed-scale analyses that are designed to address questions about spatial structure and diversity are in early stages of development in recovery planning. Yet, the analyses described above can contribute some of the information needed to set goals related to spatial structure and diversity. In particular, both coarse and fine-scale (see Table 3-1) comparisons of current and historic habitat within a watershed can indicate whether the kinds of habitat available to salmonids today are vastly different from the past, and subsequently whether life-history types have been eliminated or vastly altered. For example, Central valley spring Chinook of Sacramento chinook are extinct from much of the ESU due to the construction of multiple dams that block access to higher elevation spawning areas. Similarly, in the Lower Columbia spring Chinook ESU, historical populations in the upper Colltiz, Cispu, Tilton, Toutle, Kalama, Lewis, Sandy, Big White Salmon and Hood Rivers have all been extirpated due to the construction of dams. Only the undammed Klama maintains a small remnant population of spring run chinook. Similar examples illustrate how habitat alterations in a watershed have resulted in shifts in spatial structure and diversity. Lake Washington is an excellent example, as it used to drain through the Duwamish River and now drains through the ship canal. This change has resulted in shifts in connectivity of populations in the Cedar, White, Duwamish, and Puyallup Rivers.

Conclusions

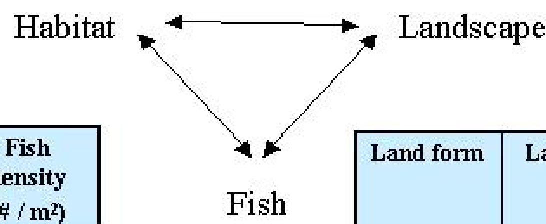
Analyses at multiple scales are necessary for setting goals related to abundance, productivity, spatial structure and diversity. These large and small-scale analyses compare current and historic habitats to first evaluate whether current habitat is sufficient to support viable salmon populations and ESUs. ESU scale analyses apply consistent techniques to examine questions over large spatial scales, often relying spatial data and coarser resolution habitat data. Watershed scale analyses focus on questions relevant to individual populations.

The analyses describe in this section are designed to answer questions necessary to set biological recovery goals (Phase I). As these recovery goals are set, the next phase of recovery planning must address how to reach those goals. The data and information collected for the analyses discussed in this section can also contribute to efforts to develop prioritized lists of actions for ESUs and watersheds. The following sections will examine these questions pertinent to Phase II recovery planning in more detail.

*****FOR REVIEW ONLY*****

a.

		Habitat type			
		Main stem		Tributary	
		Distribution	Characteristics	Distribution	Characteristics
Land use category	Urban	20%	% riffles	10%	% lost
	Forested	30 %	% riffles	60%	% lost
	Agriculture	50 %	% riffles	30 %	% lost
	Total	100%		100%	



c.

Habitat type	Fish density (# / m²)
Main stem	12
Tributary	15
Estuary	20

b.

Land form	Land use	Fish density (# / m²)
Erosive geology	Urban Forested Ag	12 20 5
Non-erosive geology	Urban Forested Ag	15 2 8

Figure 3-1. Illustration of the kinds of relationships between habitats, landscape, and fish that broad-scale analyses can examine: Table A illustrates how habitat quantity and characteristics are distributed across different land uses; Table B illustrates how fish density in three land use categories differ in erosive vs. non-erosive geologic settings; and Table C describes how fish density differs across habitat types. The numbers in the tables are completely fictitious and for illustrative purposes only.

*****FOR REVIEW ONLY*****

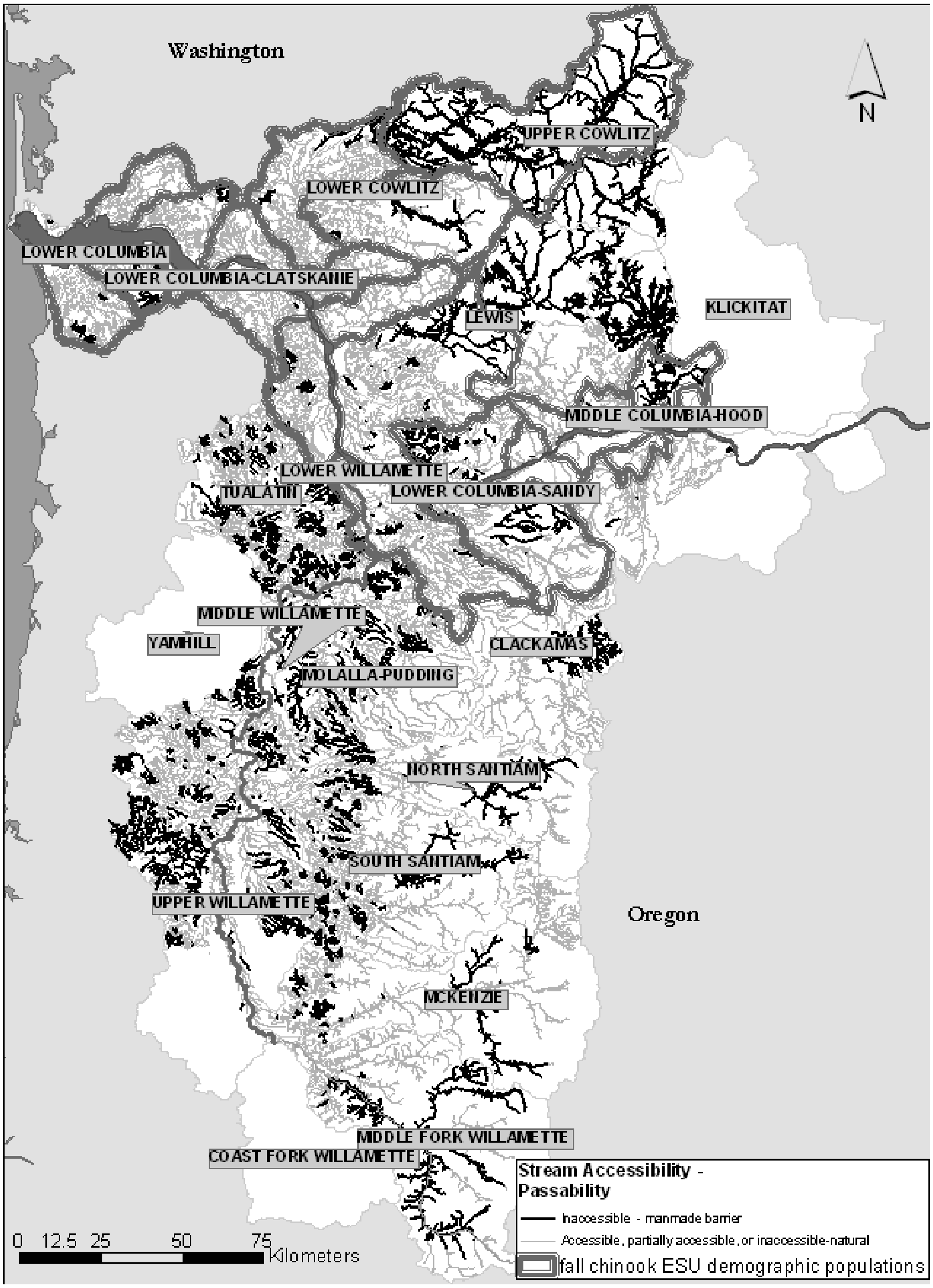


Figure 3-2. Stream accessibility and passability for all streams considered in the WLC analysis. Legend describes categories of accessibility. Stream km that are inaccessible because of manmade barriers are indicated in black. Labels indicate the 4th field hydrologic basin, and thick gray outlines indicate the boundaries fall chinook demographic populations for the Lower Columbia fall chinook ESU.

*****FOR REVIEW ONLY*****

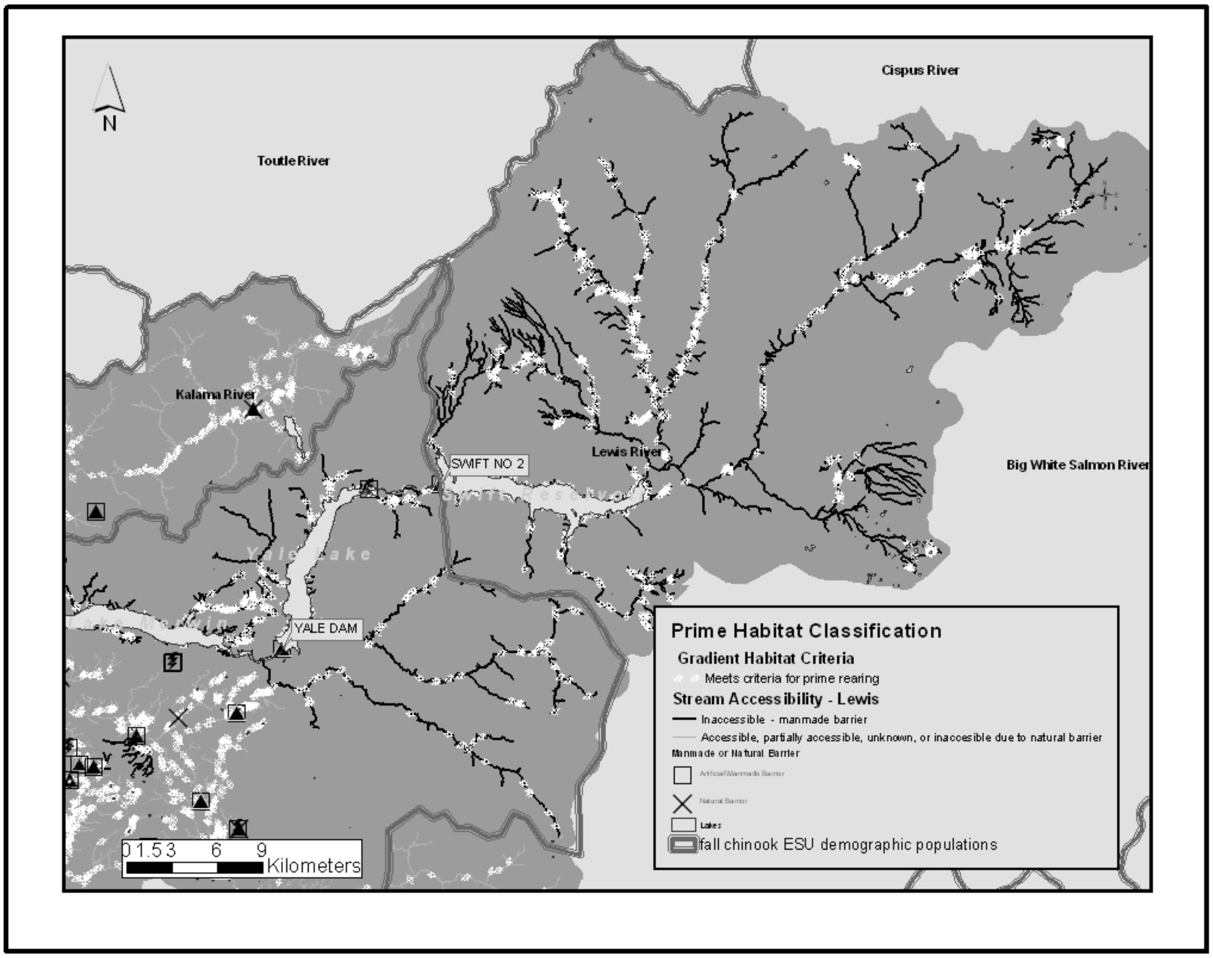


Figure 3-3. Example of the identification of prime and possible habitat attributes. Map indicates stream reaches classified as “prime habitat” for chinook rearing or spawning in the Lewis River, based only on defined gradient thresholds. The white symbols indicate patches of streams (reaches) that meet the thresholds. Black lines represent streams inaccessible to fish due to manmade barriers. Thick gray lines represent the boundaries of fall chinook demographic populations included in the Lower Columbia fall chinook ESU.

*****FOR REVIEW ONLY*****

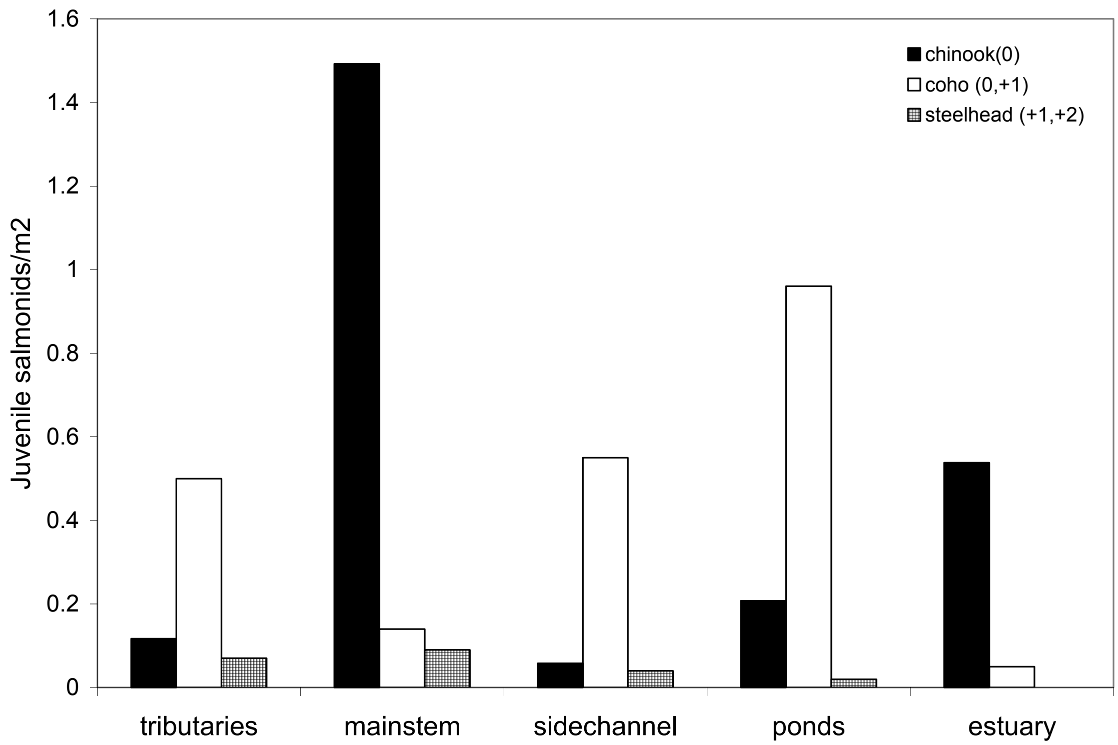


Figure 3-4. General juvenile salmonid use at the habitat scale. Compilation of over 60 references.

*****FOR REVIEW ONLY*****

Table 3-1. Habitat types used for the two types of watershed assessments described in this paper. Coarser-scale habitat types are mapped from topographic maps, aerial photography, and satellite imagery. Finer-scale habitat types are mapped using a combination of aerial photography (for larger units) and field measurements. Adapted from Beechie et al. (in press).

Habitat type (Coarser scale) ◄	———	► **Habitat type (Finer scale)**
Large main stems (>50m bfw) by channel type based on gradient and confinement	• Mid-channel • Edge	• Mid-channel pool • Mid-channel glide • Mid-channel riffle Boulder/cobble Cobble/gravel • Bar edge • Bank edge Natural Hardened • Backwater (alcove)
Small main stems (10-50m bfw) and tributaries (<10m bfw) by channel type based on gradient and confinement	• Pools • Riffles	• Pool Scour Plunge Trench Backwater • Glide • Run • Rapid • Riffle
Off-channel habitat within large main channel floodplains	• Channel-like • Pond-like	
Impoundments		• Ponds < 500 m^2 • Ponds > 500 m^2 and < 5ha • Lakes > 5 ha
Palustrine wetland	• Forested • Scrub/shrub	Open water area by season
Riverine tidal wetland	• Forested • Scrub/shrub	Open water area by season and tidal stage
Tidal-delta wetland	• Scrub/shrub • Emergent	Open water area by season and tidal stage
Tidal-delta channel	• Mainstem • Blind • Distributary	

*****FOR REVIEW ONLY*****

Table 3-2. List of analyses that address questions pertaining to the four categories of recovery goals for salmon.

Recovery goal	Examples of analyses	References
Abundance and productivity	Quantification of current vs. historic habitats	McIntosh et al. 2000, Thompson and Lee 2000, Thurow et al. [year?]
	Effects of irrigation diversions in the Salmon River Basin	McClure et. al (in prep.)
	Land-use impacts on salmon	Bradford and Irvine 2000, Thompson and Lee 2000, Paulsen and Fisher 2001
Diversity and spatial structure	Quantification of current vs. historic habitats	Thurow et al. [year?]
	Regional patterns in fish diversity	Frissell et al. 1993, Waples et al. 2002
	Associations between fish assemblages and habitat	Waite and Carpenter 2000
	Presence/absence mapping	Quigley and Arbelbide 1997

Table 3-3. General description of potential analytical approaches for estimating historical and current habitat abundance at the ESU level. Note that methods are not readily available for estimating historical conditions for most habitat types. Thus, ESU-scale analyses most commonly focus on direct loss of important habitat types.

Type of habitat change	Possible analysis methods
Tributary and mainstem blockages	Utilize digital information on known barriers and historical ranges (e.g., Quigley and Arbelbide 1997). May also model historical ranges based on channel slopes and stream size [reference?].
Channel type (e.g., based on Montgomery and Buffington 1997)	Estimated from vegetation information, hydrography and digital elevation models (Lunetta et al. 1997).
Off-channel, wetland or beaver pond areas	No remote sensing methods readily available.
Lakes	No remote sensing method available.
Beaver ponds	No remote sensing method available.
Estuary	No remote sensing method available. Use existing information where possible (e.g., Bortelson 1980).

*****FOR REVIEW ONLY*****

Table 3-4. General description of analysis approaches for estimating historical and current habitat abundance at the watershed-level. References provide more detail on specifics of methods.

Habitat type	Analysis methods	References
Reduced off-channel or wetland areas	Historical habitat areas estimated from historical maps, notes, and photos, and often field verified by residual evidence of their prior locations. Present-day areas can be measured from aerial photographs and in the field.	Beechie et al. 1994, Collins and Montgomery 2001.
Lakes	Changes to lake areas are measured directly from historical and current maps and typically indicate where rivers have been dammed for hydropower or water supplies.	Beechie et al. 1994
Beaver ponds	Pre-settlement beaver pond areas estimated based on frequencies of beaver ponds in relatively pristine areas, or predictive methods using stream and valley characteristics. Present-day pond areas within the study area measured using field surveys and aerial photography.	Naiman et al. 1988, Pollock and Pess 1998
Tributary and mainstem blockages	Portions of tributaries that are no longer accessible to salmon can be mapped using inventories of habitat upstream of migration barriers. Natural barriers to salmon migration must first be identified to delineate the assessment area. Habitat areas upstream of each manmade barrier must be surveyed to determine how much habitat is inaccessible.	Beechie et al. 1994, WDFW 1998, OWEB, Pess example
Altered pool abundance	Based primarily on data from reference sites within the study area, but may also use historical information where available.	Beechie et al. 1994, WDFW 1998, OWEB, N-L model, Collins and Montgomery 2001, and Eastside pools study

HABITAT ANALYSES FOR PHASE II RECOVERY PLANNING: IDENTIFYING ECOSYSTEM RESTORATION ACTIONS

George R. Pess, Timothy J. Beechie, Sarah A. Morley, and Eric M. Beamer

In this section we describe a number of inventories and assessments that help identify ecosystem restoration actions. These assessments focus on how land uses have disrupted ecosystem processes and functions that support salmon. There are two types of inventories that can be used to help identify ecosystem disruptions: identifying altered ecosystem processes and identifying impaired biological integrity (see also Figure 2-1). The first group of inventories describes historical and current ecosystem processes and functions (e.g., sediment supply, riparian functions, habitat connectivity) in order to determine which of them have been disrupted. This set of inventories generally leads directly to lists of ecosystem recovery actions. The second set of inventories identifies locations where biological integrity has been impaired, and can indicate potential ecosystem disturbances. These inventories typically do not directly identify restoration actions, but can indicate which ecosystem processes are disrupted and therefore which process assessments should be conducted to identify recovery actions. In general terms, both types of inventories are relevant in all ecoregions (see “An assessment Approach for Habitat Recovery Planning” section for ecoregion descriptions), although specifics of the assessments and inventories may vary depending on which ecosystem processes are most important in different ecoregions.

Assessing Degradation of Ecosystem Processes and Functions

In the "An assessment Approach for Habitat Recovery Planning" section we presented a conceptual diagram illustrating how watershed controls and natural landscape processes combine to form habitat conditions. Here we modify the conceptual diagram to focus on two processes for the purpose of explaining linkages among ultimate controls, proximate controls, habitat-forming processes, habitat conditions, and biological responses (Figure 4-1). Ultimate controls are independent of land management over the long term (centuries to millennia), act over large areas (>1 km^2), and shape the range of possible processes and habitat conditions in a watershed (Naiman et al. 1992, Beechie and Bolton 1999). Proximate controls are affected by land management over the short term (i.e., years to decades), and act over smaller areas (Naiman et al. 1992). Different watershed controls and natural landscape processes will dominate in different ecoregions, leading to differences in the assessments to be conducted for identifying ecosystem restoration actions (Table 2-3). For example, the Puget Lowlands may be dominated by frequent winter flooding which can lead to large inputs of sediment from landslides on an annual basis, while the Eastern Cascades or Blue Mountains of Oregon may be dominated by fire that has a periodicity between 10 to 20 years, which can lead to large-scale gully erosion (Olson 2000). In each case sediment is the input, but the process and rate is different, which can lead to a different habitat response over the long-term.

Landscape processes are typically measured as rates and characterize what ecosystems or components of ecosystems do (SWC 1998). For example, sediment or hydrologic processes in a watershed may be characterized by the rates (volume/area/time) at which sediment or water is supplied to and transported through specific locations of a watershed. Certain riparian functions can be viewed similarly. For example, wood recruitment to streams from riparian forests and wood depletion from the channel are both rate functions. Natural rates of landscape processes are here defined as those that existed prior to non-Native American settlement and development activities, mainly forestry, agriculture, and rural and urban residential and commercial development.

Ecosystem processes and functions to inventory should include (at a minimum) hydrology, sediment supply, riparian functions, channel-floodplain interactions, habitat isolated from salmon access, and water quality (Table 4-1). This suite of inventories is based on current scientific knowledge of their effects on salmonid habitat and survival of salmon in freshwater, as well as knowledge of how various land use practices affect the processes. The list may not include all impacts to salmon in a watershed, but it includes those that are clearly supported by scientific literature (e.g., Meehan 1991, WDNR 1995, OWEB 1999a) and that are responsible for a significant proportion of the total loss in salmon production from Pacific Northwest river basins (e.g., Beechie et al. 2001). For each process a series of diagnostics and inventories can be developed based on rates from scientific literature and local studies (e.g., Appendix B).

These inventories identify: 1) the natural landscape processes active in a watershed, 2) the effects of land use on natural processes, and 3) the causal relationships between land use and habitat conditions. Habitat restoration and protection actions resulting from these assessments are directed at protecting and restoring beneficial habitat-forming processes instead of attempting to build specific habitat conditions (FEMAT 1993, Spence et al. 1996, Moore 1997, Beechie and Bolton 1999). These assessments systematically identify land use disruptions to habitat-forming processes at two levels of resolution. Coarser level assessments locate disturbed habitat-forming processes using a combination of historic stream flow data, Geographic Information System

(GIS) data (e.g., Lunetta et al. 1997, Quigley and Abelbide 1997), and field-based inventories to identify alterations to the flow regime, sediment supply, riparian conditions, blockages to salmon migration, water quality, and channel and floodplain interactions (e.g., WDFW 1998). These assessments provide broad-brush tools for understanding where processes are disrupted, and in some cases for estimating total costs of restoration actions. The finer level assessment relies solely on field-based inventories and identifies specific restoration or protection actions that are required for recovery.

Inventories of ecosystem processes and functions can be grouped into three categories: 1) distributed watershed processes (similar to non-point sources, such as supplies of sediment and water), 2) reach-level processes that primarily affect the adjacent reach (e.g., riparian functions), and 3) other ecosystem functions. The inventory of disruptions to distributed watershed processes can have at least two levels of resolution: a coarse resolution that identifies areas within a watershed where land uses have increased rates above natural background levels, and a detailed field inventory that identifies specific areas where restoration actions are needed. Inventories of disruptions to reach-level processes can also have two levels of resolution: a coarse resolution analysis that broadly indicates patterns of degradation in the watershed, and a detailed field inventory that identifies site-specific disruptions to reach-level processes and potential restoration actions. Other ecosystem functions are necessary attributes of ecosystems that are not readily analyzed as rates or levels of function. For example, barrier inventories describe where habitats have been blocked to salmon access, and flow diversion and screening inventories describe mortality of fish diverted or pumped into irrigation systems. Neither type of assessment neatly fits into the categories of watershed-level or reach-level processes.

Ecosystem recovery actions for watershed processes can be grouped into habitat protection actions and habitat restoration actions. Habitat protection actions are those actions that are intended conserve existing functioning ecosystem processes. Habitat restoration actions are those intended to restore impaired ecosystem processes or functions. Both are identified from the assessments of disrupted processes, with areas not disrupted generally needing protection, and areas of disrupted processes generally needing restoration (e.g., Appendix B). Restoration actions can be furthered subdivided in active and passive restoration. Passive restoration actions are those where a land use is removed or altered in such a way that a process recovers naturally. Active restoration projects are those where specific interventions are used to assist recovery of a watershed process.

Watershed-level Processes

Watershed level processes are those that have multiple, widely distributed sources, including sediment supply, hydrology, and inputs of nutrients or pesticides (Table 4-1). Describing how these processes have been disrupted and what restoration actions are required for their recovery requires two different kinds of assessments. First, process assessments identify the degree to which process has been altered by land use, and where in each watershed these changes have occurred. Second, inventories are required to identify where specific restoration actions must be taken in order for recovery to occur.

Identifying altered sediment supply

Sediment supply to streams is altered by many processes in including changes in mass wasting due to logging and road building (e.g., Sidle et al. 1985), increased surface erosion after prescribed burns (e.g., Megahan et al. 1995), increased surface erosion from unpaved road surfaces (e.g., Bilby et al. 1989), and surface erosion and gullying after grazing (Platts 1991, Elmore 1992, Johnson 1992, Trimble and Mendel 1995). We present two main approaches to understanding disruptions to these sediment supply processes: (1) budgeting and (2) landscape indicators based on known relationships of certain water and land uses to the parameter in question. The budgeting approach is often used for sediment supply, but can also be used for inputs of nutrients or pesticides to water bodies. The general budget can be stated in equation form:

$$\Delta S = I - O$$

Where, ΔS is change in storage, I is input, and O is output (e.g., Reid and Dunne 1995). In essence, S is the stream condition for any parameter (e.g., the amount of sediment or of a pesticide in the stream), and quantifying changes in inputs or outputs indicates how land uses have altered the stream ecosystem. In many cases, it may only be necessary to quantify how inputs have been altered by land uses, which is called a partial budget. That is, where changes to outputs are negligible, an increased input is approximately equal to the change in storage and to the altered stream condition. Therefore, it is not necessary to understand output processes in detail (e.g., sediment transport) in order calculate change in storage and to understand how the stream ecosystem has been altered.

We illustrate the partial budget approach by describing how one might approach two different altered sediment inputs to streams due to land uses: 1) extrapolation of limited empirical data for estimating increased fine sediment supply from tilled croplands in the Blue Mountain Level III Ecoregion, and 2) an empirical approach to estimating changes in landslide rates due to forestry activities in the North Cascades Level III Ecoregion. Both approaches focus on identifying where sediment supplies to streams have been significantly altered, and can help focus restoration efforts on areas that contribute large amounts of sediment. Note that these approaches do not identify the exact locations of necessary restoration actions, which may require specific inventories of croplands where eroded sediments deliver directly to streams or road segments with high risk of landsliding. For efficiency, these inventories can initially target those areas of high sediment supply identified by the partial sediment budgets.

The partial sediment budget for croplands makes use of measured erosion rates from soils with and without cover crop. Soils without cover crop erode at rates as much as 10 times higher than that of soils with cover crop (see complete overview of processes and rates in Dunne and Leopold 1978), and varies with soil types, rainfall, slope, cover type, and other factors. Local erosion rates for different soils and cover crops may have been measured in many areas by researchers and land management agencies, and examples for different soils and cover crops can be found in Dunne and Leopold (1978). From known rates, a simple cumulative model for basins with can be expressed as

$$I_c = (A_c \times E_c) + (A_{nc} \times E_{nc})$$

Where I_c is the current total estimated sediment input to selected reach, A_c is the area of land with cover crop or other vegetation, E_c is the erosion rate per unit area with cover crop, A_{nc} is the area of land with no cover crop, and E_{nc} is the erosion rate per unit area without cover crop. A natural background rate of sediment supply from surface erosion can be estimated by applying the natural erosion rate to the entire basin:

$$I_n = (A_c + A_{nc}) \times E_{nc}$$

Calculations of I_c/I_n for various subwatersheds within a region can then be compared to identify those areas where tilling is likely to have significantly altered sediment supply to streams.

Where soil erosion rates have not been measured, it may be necessary to use predictive equations such as the Universal Soil Loss Equation (Wischmeier and Smith 1965, Wischmeier and Smith 1978) or Revised Universal Soil Loss Equation (Renard et al. 1991, Renard et al. 1997). Dunne and Leopold (1978) provides a good overview of the equation and its application, along with charts and tables for estimating certain parameters in the equation, and the original handbooks can be consulted for greater detail on the methods. Estimates of local soil parameters are generally available from local Soil Conservation Service offices.

In the second example, a partial budget for sediment supply in coastal forests is constructed by conducting landslide inventories from historical aerial photographs and estimating contributions of fine sediments from road surface erosion (e.g., Paulson 1997). The general approach is similar to that for the cropland erosion example, where the final product of the partial sediment budget is I_c/I_n and comparisons among basins indicate areas where sediment supplies have been altered significantly. In these inventories landslides are enumerated and measured on each aerial photograph, and volume of each landslide is calculated based on a relationship of photo-measured area to field-measured volume for a subset of the recent landslides. Land use association is also recorded for each landslide (e.g., clear-cut, road, or mature forest), allowing estimation of the aggregate impact of land use on the sediment input, as well as identification of the land uses most responsible for changes in sediment supply. Surface erosion estimates can be based on characteristics of road surfaces, cut and fill slopes, and precipitation (e.g., WDNR 1995). Calculation of I_n typically assumes that there is no surface erosion (overland flow and gullying are rare in the Coastal Forest ecoregion), and the landslide sediment production rate for mature forests can be applied to the entire basin. These sediment budgets can then be compared to identify areas where sediment supplies have been most altered and where modified timber harvest practices or road modifications can have the greatest impact on ecosystem recovery.

Landscape and land use indicators of altered sediment supplies can be developed from sediment budgets or field studies of erosion in order to more rapidly screen large areas for disrupted sediment supply. For example, GIS maps of geology, soils and hillslope angles can identify areas that are prone to landsliding (e.g., Montgomery and Dietrich 1994, Appendix B), and overlays of land cover and roads can be used to identify where landsliding has likely increased (Appendix B). Results of such screening analyses can then be used to identify areas where inventories of potential restoration actions should be focused.

In general, remote sensing methods and mapping of landscape indicators are used to identify areas for passive restoration, and field inventories are required to identify active restoration projects. For mass wasting dominated areas, mapping of landslide hazard areas is used to identify areas that are particularly prone to landsliding, and sensitive to land uses such as clear-cut logging or road building (Figure 4-2). Such maps identify passive restoration actions (e.g., areas to avoid for future logging), which allow recovery of sediment supply rates by preventing or modifying land uses within hazard areas. Second, inventory of road landslide hazards identifies specific areas for active restoration. Road inventories should identify segments of road that are at risk of failure (e.g., Renison 1998), as well as specific stream crossings, cross drains, or fills that are likely to fail. Each potential failure site can be itemized on project lists for restoration action. These restoration actions can then be prioritized based on protection of refugia, potential impact to stream habitat, smolt production, cost, and other factors (see the “Updating the Recovery Plan” subsection, page XX).

Where surface erosion and gullying are dominant processes, terrain and soil conditions that lead to severe erosion can also identify areas where modified agricultural practices can reduce erosion and sediment supply to streams. Modeling of altered sediment yields by various conservation practices can help identify the types of agricultural practices that are greatest contributors to erosion (Williamson et al. 1998) as well as those restoration actions that are most likely to be successful (Ebbert and Roe 1998). These actions can also be itemized and prioritized based on a number of factors including the magnitude of reduced erosion, costs, and other factors (see the "Updating the Recovery Plan" subsection, page XX).

Identifying altered hydrologic regime

Hydrologic processes can be altered by land uses in a variety of ways, including increased peak flows from impervious surfaces (e.g., Booth and Jackson 1997), livestock compaction (Trimble and Mendel 1995), increased peak flows from increased snow accumulation and melt (e.g., Zeimer 1981, Harr 1986, Beschta et al. 2000), and decreased peak flows or low flows from dams and withdrawals (e.g., Richter et al. 1996, Donato 1998, Spinazola 1998). Assessments of increased peak flows typically utilize landscape indicators of changes to watershed processes based on known functional relationships between land cover and peak flows (e.g., Booth and Jackson 1997, Beschta et al. 2000) (Appendix B), and may include detailed models that project changes in peak flow hydrographs as a result of land cover changes (e.g., Booth and Jackson 1997). Assessments of low flow changes typically include inventories of total withdrawals and calculation of the proportion of stream flow removed (e.g., Donato 1998, Spinazola 1998), as well as indirect estimates based on power consumption at pumping stations (e.g., Maupin 1999).

Known relationships among zoning, impervious surface area, and changes in hydrologic processes and biota have been used in indicate changes hydrologic regime in urban areas (see Appendix B). Where impervious surface areas are less than 3% of the watershed area, hydrologic regime is not significantly different from one with no impervious surfaces (Booth and Jackson 1997). However, where impervious surfaces are more than 10% of the watershed area, hydrologic regime has likely been altered to the point where changes in biota are severe (Luchetti and Furstenburg 1992, Moscrip and Montgomery 1997, May et al. 1997). Similar coarse analyses can be developed for other watershed processes such as contaminant runoff from agricultural or urban areas, where high concentrations of compounds that are toxic to biota, such as salmon, alter their behavior in ways that could reduce survival (Scholz et al. 2000). More general landscape indicators, such as percent of a watershed in urban land cover, may be a greater predictor of biological condition because it is a more inclusive measure of anthropogenic disturbance and not just a change in one specific watershed process such as hydrologic regime or water quality (Karr and Chu 2000, Morley and Karr 2001). Examples of use of land cover indices for peak flow changes from rain on snow are also included in Appendix B.

Indicators of hydrologic alteration (IHA) can be used to assess the degree of hydrologic alteration within a watershed (Ritcher et al. 1996). The method summarizes complex hydrologic variation using 32 stream flow parameters that have biologically relevant attributes (Ritcher et al. 1996). The hydrologic data is from known sources such as stream gages and wells. Each parameter is broken up into pre and post impact time frames, and the central tendency (defined as the mean and median) and dispersion (defined as the variance and coefficient of variation) is compared to assess degree of hydrologic perturbation (Ritcher et al. 1996). Perturbation can include activities such as dam operations, flow diversion, groundwater pumping, or intensive land-use (Ritcher et al. 1997). The tool is to be used with other ecosystem metrics (e.g., biological integrity indices) and can help set stream flow based restoration targets, identify areas of hydrologic alteration, and measure progress towards quantified conservation goals (Ritcher et al. 1996, 1997, 1998). The tool is not meant to predict biological response to hydrologic alteration (Ritcher et al. 1996).

For alteration of low flows by water storage and diversions, data availability varies between large dams and small private irrigation or water supply diversions (Spence et al. 1996, Quigley and Abelbide 1997). For large dams (greater than 2 m in height) inventory data are available for regional characterization of water withdrawals, and cumulative withdrawals from large dams can be calculated (e.g., Quigley and Abelbide 1997). However, data for smaller diversions and their effect on stream flows are less readily available (Spence et al. 1996), and inventories of low flow changes are needed to systematically identify stream reaches where low flows are at issue within individual watersheds.

Two types of assessments can be used to inventory low-flow impairments within watersheds and identify potential restoration actions. First, low flow impairments can be identified through inventories of diversions and quantification of water withdrawals. The existing 303d database from EPA identifies more than 13,000 km of flow-impaired stream reaches in the interior Columbia basin (Quigley and Abelbide 1997) as well as additional streams in western Washington and Oregon. This database can be used as a preliminary inventory of withdrawals and low flow impairments. More detailed field inventories of withdrawals can be conducted within watersheds to list all small dams, diversions, and pump stations that withdraw water from streams and assess the degree to which stream flows are reduced. State water rights data bases (e.g., Washington Department of Ecology, Oregon Water Resources Department) provide a starting point for such assessments, although field inventories of actual withdrawals are often needed to confirm which water rights are currently in use. In general, ranking of the proportion of stream flow withdrawn in various reaches indicates which reaches deviate most significantly from natural stream flows, and are most likely in need of increased stream flows. Methods available to identify specific locations where water withdrawals impact stream flows include source metering using stream flow gauges or other measurement devices (e.g., Donato 1998), direct assessment of stream flow change (e.g., Ritcher et al. 1996), and estimates of water withdrawal based on power consumption at pumping stations (e.g., Maupin 1995).

In a similar fashion, identifying recovery actions for alterations to peak flow hydrology include both protection and restoration actions. In areas where hydrologic regime approximates the natural regime, ecosystem management should focus on protecting current hydrologic processes. These actions might include avoiding additional hydrologic changes by preventing new impervious services and forestry impacts to peak flows. By contrast, where hydrologic regime deviates significantly from the natural regime natural, restoration actions should be identified. For alterations to peak flows, identification of restoration actions may include actions to alleviate impervious surfaces (Holz et al. 1998, Maryland DER 2000) and to alleviate impacts of clear-cuts and roads on peak flow responses. For low flow impairment, identification of restoration actions is perhaps more problematic as many withdrawals are unlikely to stop completely. IFIM methods or variations of it, as well as flood recurrence information can be used to help identify how much stream flow is necessary to support aquatic ecosystems at both low flow and high flow periods (Jowett 1997).

Identifying altered water quality

Water quality parameters can also be used to indicate areas with a high likelihood of disruption, especially with regard to temperature and nutrient or pesticide inputs. Again, the EPA 303d list provides a useful starting point for identifying disruptions to water quality. Many streams throughout the west are listed as water quality impaired, which indicates that some type of restoration may be necessary. In general, further field inventories may be necessary to clarify the exact nature of the problem, and then to identify corrective actions.

Water quality assessments may include direct measures of water quality parameters, and relationships among the parameters and biotic assemblages may help identify where disruptions are most important (Waite and Carpenter 2000). For example, multivariate classification and ordination were used to examine patterns in chemical and physical variables in association with relative fish relative abundance in the Willamette River Basin (Waite and Carpenter 2000). Patterns of fish assemblages were primarily related to water temperature, dissolved oxygen, and stream channel gradient at the ecoregion scale, however, chemical concentrations of pesticides and total phosphorus were more important than physical habitat features in low gradient floodplain ecoregions such as the Willamette Valley (Waite and Carpenter 2000). Water quality restoration actions may thus need to be a priority in such areas. (Additional biological indicators are discussed later in this section.)

Reach-level Processes

Reach-level processes are those processes that directly affect the adjacent reach (Table 4-1). These processes mainly include riparian functions and floodplain-channel interactions. An extensive body of literature describes linkages between riparian forest functions and stream habitat, which in turn affect the productivity and abundance of salmonids. Riparian functions include supply of wood and leaf litter to streams (Naiman et al. 1992), shading (Beschta et al. 1987), root reinforcement of stream banks and floodplain soils (Platts 1991, Elmore 1992). Dominant functions vary by ecoregion, although many streams even in the driest ecoregions have or had a forested riparian corridor (Platts 1991). Channel and floodplain interactions form a wide array of habitats that salmonids historically occupied (Sedell and Luchessa 1982, Peterson and Reid 1984, Collins et al. 2000). Many of these habitats are now either destroyed or inaccessible to salmon due to the effects of levees, dams, channel incision, or other land uses (Sedell and Luchessa 1982, Beechie et al. 1994, Peacock 1994, Shafroth 1999, Pohl 1999, Beechie et al. 2001, Collins and Montgomery 2002).

Identifying disrupted riparian processes

The level of wood input or other riparian functions increases with increasing width of forest buffer on streams (Figure 4-3), and the proportion of the function occurring within a given distance of the channel edge varies by function (e.g., Sedell et al. 1997). These types of relationships can be used to evaluate the current status of functional interaction between a stream reach and riparian area, and indicate whether existing levels of riparian protection are sufficient to ensure continued function. For this example we focus on recruitment of wood to streams for forested riparian zones (Murphy and Koski 1989, Carlson et al. 1990, Van Sickle and Gregory 1990, Beechie et al. 2001, Wing and Skaugset 2002) and its function in channels (e.g., Bilby and Ward 1991, Montgomery et al. 1995, Abbe and Montgomery 1996, Beechie and Sibley 1997, Collins et al. 2002), which are among the most studied of riparian functions. However, a similar assessment approach can be applied to other types of riparian systems and for different riparian functions.

As with watershed-level processes, there are two types of assessments required for reach-level processes. The first assessment identifies where reach level processes have been disrupted. The distribution of riparian conditions at this larger spatial scale can provide a general sense of the change in riparian function from historic conditions (e.g., Lunetta et al. 1997, OWEB 1999a). Subwatersheds where the current distribution of riparian conditions deviates markedly from that expected under a natural disturbance regime are locations where riparian restoration efforts may be appropriate. The same data can also help managers understand how different land use practices differ in their degree of impact on riparian functions. These relationships can then help assess the potential impacts of large-scale land use policies on salmon habitat recovery (e.g., evaluating potential effects of growth management legislation).

Because of limitations in the satellite classification of riparian forests, field inventories of riparian sites must be used to identify specific restoration actions (Clary and Leninger 2000, see Appendix B). Field inventories may consist of initial measurements and classification from aerial photography, combined with field confirmation of the riparian vegetation conditions for each stream reach. Newly developed multispectral technologies may also be of use in identifying riparian conditions with sufficient detail for site-scale planning purposes. At a minimum, they should classify riparian conditions by buffer width, stand type, and age of vegetation. From the data, managers can identify impaired or moderately impaired stream segments in order to determine the likely cause of that impairment, and identify required restoration actions. In general, impairment is defined with respect to a natural reference condition for the area in question, which is usually based on historical information (e.g., OWEB

1999a, Collins and Montgomery 2000, Appendix B). However, in grazed riparian areas where the natural riparian vegetation is not forested, identification of disrupted riparian function may rely on measures of stubble height as an indicator of the level of disturbance (Clary and Leninger 2000, Turner and Clary 2001).

Regardless of current condition of riparian areas, establishing protected areas along the channel where natural riparian vegetation can develop through time and interact with the stream is a necessary component of riparian restoration. Active restoration efforts may be appropriate at currently impaired sites. Riparian restoration may include exclusion of livestock in drier riparian systems with less woody vegetation (Clary et al. 1996, Clary 1999), as well as the planting of desired riparian plant species or manipulation of the existing vegetation to accelerate tree growth and the development of desired stand structural characteristics (Berg et al. 1996, Beechie et al. 2000). Inventories of disrupted riparian functions in drier ecoregions are conceptually similar, but historical and present-day disrupted riparian communities will differ (e.g., the use of ecoregion information in OWEB 1999a).

Channel-floodplain interactions

Disruptions to floodplain and channel interactions may also dramatically reduce abundance of wood, pools, and off-channel habitats in larger river systems. These disruptions may result from altered sediment and wood supplies (e.g., downstream of dams) (Pohl 1999, Shafroth 1999), installation of dikes and riprap to control channel movement (e.g., Beechie et al. 1994, Collins and Montgomery 2002), and channel incision that isolates a channel from its floodplain (e.g., Peacock 1994). Inventories that can help identify these disruptions include measurement of channels no-longer accessible to salmon (Sedell and Luchessa 1982, Beechie et al. 1994), mapping of dikes, riprap, and disconnected floodplain surfaces (Appendix B), aerial photograph inventories of channel and habitat changes downstream of dams (Pohl 1999, Shafroth 1999), and inventories of incised channel segments.

Stream channel classification systems are another inventory tool that can be used to help identify disruptions in channel-floodplain interactions. Stream classification systems use the same suite of key variables such as geology, valley floor constraint, and channel slope to determine stream channel type and response of channels to changes in inputs such as water, wood, sediment, and energy over an entire watershed (Table 4-2). Stream channel classification systems reduces the number of variables and measurements needed to differentiate site response to channel-floodplain disruption, allows for the classification of spatial variability by grouping channels across a watershed, and creates a consistent method that can be used on a larger-scale across watersheds. The features used to classify channel reaches and valley segments by identifying common characteristics or patterns are often relevant for the development of channel-floodplain restoration plans, as stream reaches that have similar physical characteristics will respond to restoration actions similarly. These classification systems should be used in conjunction with the preceding inventory methods.

Habitat Connectivity—Impaired Fish Passage

Stream crossing structures that block fish access to useable habitats has been a recognized problem for many years in the Northwest. Such blockages can account for as much as 50% of lost smolt production from tributaries in Puget Sound river basins (Beechie et al. 1994, Beechie et al. 2001). Since 1992, the Washington Department of Transportation (WSDOT) has examined 3,216 fish crossings on state roads and documented 1,556 fish barriers (Johnson et al.

2001). Forest landowners, state agencies, and federal agencies in the state of Oregon have identified almost 3,000 culverts on fish bearing streams since 1997 (The Oregon Plan for Salmon and Watersheds 2001). Private landowners, local, state, and federal agencies, and watershed groups have recognized magnitude of this problem for the last decade and have developed systematic methods to fix the problem through barrier inventory, assessment, and allocation of funds to correct the fish passage problems identified (ODFW 2001, WDFW 2001). For example, since 1995 over 900 locations throughout Oregon have been made more accessible to fish (The Oregon Plan for Salmon and Watersheds 2001). Because there is not one comprehensive inventory database for all fish blockages in each state or throughout the Northwest, it is difficult to document the entire picture of what has been accomplished to date or the overall benefit to fish populations (The Oregon Plan for Salmon and Watersheds 2001). Nevertheless this is an important component of restoration that has been ongoing and is needed for the recovery of Pacific Northwest salmon populations.

Assessing such isolation of habitats is one of the simplest inventories that can be conducted because criteria for fish migration blockages are relatively clear and identifying the amount of habitat affected involves little subjectivity. All the Northwest states have developed fish passage criteria for juvenile and adult salmonids that can be used as the basis for identifying fish blockages (ODFW 2001, WDFW 2001). Moreover, combining these inventory results with cost estimates for restoration actions allows managers to rank the cost-effectiveness of individual projects in order to more effectively direct the expenditure of limited restoration funds.

For example, Snohomish County Surface Water Management (SCSWM) combined eight inventories that identify isolated habitat in the Stillaguamish River Basin from a variety of sources including SCSWM, Washington Department of Fish and Wildlife (WDFW), the

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Washington State Department of Natural Resources, the Stillaguamish Tribe, and the U.S. Forest Service (Mike Purser, SCSWM, personal communication) (Figure 4-4). Three out of the five agencies doing the inventories followed the Fish Passage Barrier and Prioritization Manual of Washington Department of Fish and Wildlife (WDFW 1998). Based on an inventory of 952 structures as of 2002, they identified 544 structures as 100% passable to salmonids, 337 structures as less than 100% passable, and 71 structures that as unknown due to no information (Mike Purser, SCSWM, personal communication).

One of the inventories has been used to evaluate the cost-effectiveness of reconnection projects based on the habitat area upstream of the project, multiplied by the average life span of a blockage (~50 years) and divided by the cost of the project (Pess et al. in press). These results allowed natural resource agencies to identify the most cost effective projects for reconnecting blocked tributary habitats based on benefits to multiple salmonid species, as well as costs of reconstructing individual stream crossings (Pess et al. in press).

Water diversions can also impair fish passage, and have been a recognized problem for salmonids in the Northwest. There are almost 76,000 permitted water diversions in Oregon alone, however, many of these do not affect ESA-listed salmonids (The Oregon Plan for Salmon and Watersheds 2001). Less than 1,000 of these diversions are required to be screened because they are greater than 30 cubic feet/second (cfs), however only five have been screened to date (The Oregon Plan for Salmon and Watersheds 2001). NMFS, California, Idaho, Oregon, and Washington have all developed criteria to exclude both juvenile and adult salmonids from being entrained in water diverted without being impinged on the diversion screens (NMFS 1995, CDFG 2000, ODFW 2001, WDFW 2001). The criteria developed by federal and state agencies allows for the development of inventories because it involves less subjectivity. Again, combining these inventory results with cost estimates for restoration actions allows managers to rank the cost-effectiveness of individual projects in order to more effectively direct the expenditure of limited restoration funds.

Assessing Biological Integrity

A key component of salmon habitat is the stream biota itself. Invertebrates, amphibians, diatoms, and other stream organisms are integral parts of the aquatic food web upon which threatened and endangered fish species depend. These assemblages are also sensitive to a variety of watershed disturbances expressed over multiple spatial scales, and therefore excellent indicators of stream condition. Unlike anadromous fishes that are subject to varied disturbances in both the marine and freshwater environment (e.g., migration blockages, interaction with hatchery fish, damaged estuarine habitats, or overharvest) less migratory stream organisms often provide a more accurate reflection of site condition. In particular, much research in the field of biological assessment (measuring and evaluating biota directly) has focused upon benthic invertebrates as indicator organisms (Rosenburg and Resh 1993, Merritt and Cummins 1996). Over the past century, bioassessment techniques using invertebrates and other assemblages have ranged from saprobien indexes (Hilsenhoff 1982), to toxicity testing (Buikema and Voshell 1993), indicator species abundance (Farwell et al. 1999), diversity indexes (Wilhm and Dorris 1966), and more recently to multivariate models (Wright et al. 2000) and multimetric indexes (Davis and Simon 1995).

In the context of ecosystem recovery planning, these bioassessment tools can be used to identify high quality areas in need of protection, degraded reaches in need of restoration, and assist in identifying the specific stressors causing biological impairment (such as factors discussed above in the preceding subsections of this section). This type of information will help focus assessments of disrupted ecosystem processes on those impacts that are most biologically important (Beechie and Bolton 1999). Of late, many studies incorporating specific bioassessment tools have evaluated the relationships between these measures of in-stream biological condition and land uses/land cover patterns over multiple spatial scales (Steedman 1988, Richards et al. 1996, Allan et al.1997, Wang et al. 1997, Morley and Karr 2002). Some applications of this research include linking specific land use impacts and current condition of stream reaches, and setting realistic recovery goals given current land use patterns in particular river basins. We anticipate that these biological assessments will be most valuable in urban and agricultural areas where multiple impacts are likely to have occurred, and where the array of necessary process assessments may be prohibitively expensive without information to help prioritize them.

Multivariate Models

In this approach a predictive model is developed based on a large (≈ 200 sites) data set of reference (minimally disturbed) sites (Reynoldson et al. 2001). Using multivariate statistical analyses, references sites are matched to a set of habitat descriptors (e.g., stream order, elevation, etc.) and classified into groups. Level of impairment at a given sample site is then determined by comparison to the appropriate reference group. This approach has been most widely applied in England with the development of RIVPACS (River Invertebrate Prediction and Classification System), in Australia with AUSRIVAS (Australian River Assessment Scheme), and with BEAST (Benthic Assessment of Sediment) in Canada (Wright et al. 2000). In the Pacific Northwest, multivariate models have been developed in British Columbia for the Fraser River basin (Reynoldson et al. 2001) and in Oregon with the BORIS model (Benthic Evaluation of Oregon Rivers; Canale 1999). Based on benthic invertebrates, BORIS scores a site from 0 (severe impairment) to 100 (comparable to reference condition). A RIVPACS-type predictive model applicable to wadeable streams throughout Oregon, Washington, and Idaho is currently being developed (Hawkins and Ostermiller 2001, G. Hayslip, U.S. EPA Region 10, pers. comm., 2002).

Multimetric Indexes

Multimetric indexes, such as an index of biological integrity (IBI), integrate empirically tested attributes (metrics) of stream biota (Karr and Chu 1999). This approach was first developed using fish communities in the midwestern United States (Karr et al. 1986), but has since been modified for a variety of assemblages—most commonly fish (Simon 1998), invertebrates (Kerans and Karr 1994), and algae (Hill et al. 2000). As with multivariate models, IBIs and other multimetric indexes are regionally calibrated based on ecoregion designations and local reference conditions. In the Pacific Northwest, an IBI using benthic macroinvertebrates was developed and calibrated with data from both Oregon and Washington (Kleindl 1995, Fore et al. 1996, Morley 2000, Adams 2001). This index, known as the benthic index of biological integrity or B-IBI (Karr and Chu 1999), is composed of ten measures of taxa richness, population structure, disturbance tolerance, and feeding ecology (Table 4-3). When scores from these metrics are summed, B-IBI provides a numeric synthesis of site condition that ranges from 10 (poor) to 50 (excellent), and can determine five categories of resource condition (Doberstein et al. 2000). Multimetric indexes developed in other states differ somewhat in field methods and metrics (Hayslip 2002).

Applications

While many of the current bioassessment protocols in use today across the nation were developed largely in response to legal mandates under the Clean Water Act (CWA) passed by Congress in 1972, these monitoring tools have much application to recovery planning under the ESA. The CWA has as its main objective "to restore and maintain the chemical, physical, and biological integrity of this Nation's waters" {PL 92-500, CWA, §101(a)}. In response to this legal mandate, and under the guidance of the U.S. Environmental Protection Agency (Plafkin 1989, Barbour et al. 1999), states developed assessment protocols and water quality standards to determine if their water bodies are supporting beneficial uses such as recreation, domestic water supply, and—most pertinent to recovery planning efforts—aquatic life attainment. Although these assessment protocols and water quality standards were traditionally focused primarily on physical and chemical criteria, over the last decade there has been increasing incorporation of biological indicators which directly measure aquatic life and the maintenance of "biological integrity."

Biological integrity has been defined in many ways; we use it here as defined by Karr and Dudley (1991): "the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of the natural habitat of the regions". States and other public and private entities use information collected under Clean Water Act reporting requirements in much the same way that it could be applied to ESA recovery planning; e.g., in watershed assessments that inventory biological condition across large areas and quantify level of impairment; as a screening tool for identifying areas in need of further biological, physical, and/or chemical evaluation; in risk assessments, pollution permitting, and evaluation of proposed habitat modifications; and in prioritizing areas in need of protection and restoration, and subsequently evaluating these conservation actions (Yoder and Rankin 1998, Karr and Chu 1999, Morley and Karr 2002). In the following paragraphs, we describe biological assessment protocols currently in development or in place throughout the Pacific Northwest.

In the Western United States, both multimetric and multivariate techniques are applied by state environmental agencies to assess and report on the biological integrity of surface waters. In Idaho, the Department of Environmental Quality has developed an ecological assessment framework for both wadeable streams and larger rivers composed of four multimetric indexes based on invertebrates, fish, diatoms, and physiochemical parameters (Grafe 2000a,b). The California Department of Fish & Game uses a 19-metric invertebrate stream bioassessment procedure modified from the EPA's national Rapid Bioassessment Protocols (Barbour et al. 1999). The University of Alaska at Anchorage, working in conjunction with the state's Department of Environmental Conservation, is developing an invertebrate multimetric stream condition index for three stream types defined by gradient and substrate (Major et al. 2001). The Washington Department of Ecology and Oregon Department of Environmental Quality both currently apply a combination of multimetric and multivariate approaches to assess the condition of streams and rivers (Mochan and Mrazik 2000, Plotnikoff and Wiseman 2001). All of the states discussed above include narrative biological criteria in their water quality standards; Oregon is taking this a step further and will soon be incorporating numeric biocriteria in its standards (G. Hayslip, U.S. EPA Region 10, pers. comm., 2002).

Although it is the state environmental agencies that are largely responsible for defining, evaluating, and protecting designated uses of water bodies under the CWA, federal, tribal, and local regulatory agencies; volunteer, non-profit, and private organizations; and academic institutions also conduct bioassessments to varying degrees. At a regional level, monitoring work conducted under the Northwest Forest Plan includes bioassessment of invertebrates, periphyton, and fish (Reeves et al. 2002). In the Puget Sound region, B-IBI has been applied by city and county agencies (King County 1996, Thornburgh and Williams 2000), university scientists (May et al. 1997, Larson et al. 2001, Morley and Karr in press), and volunteers (Fore et al. 2001) to track the health of streams over time, to screen watersheds for further physical or chemical monitoring, and to evaluate various restoration and conservation strategies. For example, two studies conducted recently in Washington State used B-IBI and other invertebrate metrics to evaluate the biological effectiveness of wood placement in forested (O'Neal et al. 1999) and urban basins (Larson et al. 2001).

Tuning restoration efforts to site-specific needs is enhanced by using biology to aid in the detection of the primary causes of degradation. Both multimetric indexes and multivariate analyses provide a numeric synthesis of the biological dimensions of site condition, but they can also be broken down to derive descriptive and potentially diagnostic information from each of the component metrics. For instance, in the case of aquatic invertebrates, there are hundreds of species throughout the western states—each with specific life history requirements and varying tolerance to specific forms of disturbance (Rosenberg and Resh 1993, Merritt and Cummins 1996). The groundwork has already been laid for research on biological response signatures: "…biological community characteristics that aid in distinguishing one impact type over another" (Yoder 1991, Yoder and Rankin 1995); what remains for future research is to better link biological response variables with physical and chemical manifestations of human disturbance. Nor is bioassessment limited to lotic waters; protocols are currently being developed for lake

(Gerritsen et al. 1998), wetland (Adamus et al. 2001), and estuarine environments (Gibson et al. 2000). This evolving body of work will enable managers to better track ecological health over a larger portion of the salmon landscape.

Conclusions

Assessments of the current and historical conditions of a watershed can greatly improve our efforts to plan, implement, and monitor habitat restoration for the recovery of Pacific salmon. Systematically collected habitat data, a more thorough understanding of fish responses to habitat change, and a greater understanding of stream biota will allow refinement of the modeling tools used to predict fish and other biological response from application of different restoration strategies. These refinements will improve estimates of rates and pathways of recovery for many salmonid species in any river, and assist in prioritizing restoration actions. However, many of these refinements are still several years from completion.

In the interim, systematic inventories of disrupted habitat-forming processes and blockages to salmon migration should be conducted to provide a complete river basin overview of necessary restoration actions that can be prioritized and sequenced logically. A minimum set of inventories for any river basin should include barrier inventories, landslide inventories, floodplain and riparian characterization, channel and valley type classification, road, and biological indicator inventories. Some of these data are already available for parts of many watersheds. These data provide the basis for identifying needed restoration actions, which can be prioritized by cost-effectiveness, influence on particular species, adjacency to existing centers of biological productivity or diversity (commonly referred to as: refugia, biological hot spots, source watersheds, core areas, key habitat), or other strategies.

There are many sources of uncertainty in these assessments. Uncertainties in assessments stem from natural variability in habitat-forming processes, habitat characteristics, and fish populations, as well as from errors in assumptions and limitations of data or knowledge. Our ability to characterize these types of uncertainty is limited by availability of data on watershed processes, habitat conditions, and fish populations over long periods of time. Lack of knowledge about current habitat conditions or responses of fish populations to changing habitat conditions introduce uncertainty into predictions of fish responses to watershed and habitat restoration. Improving the quality of the data reduces uncertainty related to knowledge gaps and improves ability to address the uncertainty related to natural variability in fish response to habitat conditions.

Recovery plans designed to protect and recover processes that create and sustain riverine habitats are more likely to recover salmon of all species. Use of a comprehensive assessment process and developing restoration plans focused on the reestablishment of habitat-forming processes minimizes conflicts that can arise with species-centric restoration approaches. Restoration of habitat-forming processes targets restoration of the natural array of habitat types and conditions within a watershed, which is consistent with the concepts of watershed and ecosystem management supported by the scientific community. Moreover this approach focuses on the natural potential of each watershed, and therefore is most likely to restore the diversity and abundance of stocks appropriate to each watershed in Puget Sound.

Table 4-1. Examples of methods used for rating individual landscape processes**.**

Distributed watershed processes

Hydrology –disruption of the peak flows, low flows, and channel forming flows
A change in the magnitude, frequency, duration, and rate of minimum, mean and maximum flows can be examined using the indicators of hydrologic alteration (IHA). IHA assesses the difference in 32 biologically significant hydrologic parameters pre and post anthropogenic activities such as dam operations, flow diversion, groundwater pumping, or intensive land-use (Ritcher et al. 1997).

Hydrology - increases in peak flows
Lowland basins: Hydrologic impairment in lowland basins can be rated based on planned effective impervious area (EIA), which is the weighted average EIA upstream of the stream reach under fully developed conditions. EIA ≤ 3% is considered "functioning", EIA between 3% and 10% is "moderately impaired," and EIA > 10% is "impaired" (based on Booth and Jackson 1997, and see example in Appendix B).
Mountain basins: Peak flow ratings for mountain sub-basins can be developed based on empirical correlations between land use and elevated peak flow in forested basins (Jones and Grant 1996, and see example in Appendix B).

Sediment supply- an increase in sediment supply
Estimating impairment of sediment supply: Changes in average sediment supply for forested sub-basins within a watershed can be estimated based on present-day sediment supply rates from unlogged, clear-cut, and roaded portions of the watershed (Dietrich and Dunne 1978, Paulson 1997, Montgomery et al. 1998).
Surface erosion on agricultural and range lands: Changes in average sediment supply for croplands within a watershed can be estimated on crop practices, soil type, rainfall, slope, and other factors (Dunne and Leopold 1978).
Inventory - identify sediment reduction projects: Inventories must focus on factors that influence sediment supply, identification of landslide hazard areas so that forest practices can be avoided or modified in sensitive areas (e.g., Montgomery et al. 1998), such as risk of road-related landslides (e.g., Renison 1998), crop management practices that increase surface erosion (Wischmeier and Smith 1965), or grazing practices that alter sediment supply.

Sediment supply- a decrease in sediment supply
Routing estimates: In-channel sediment storage budget (Madej and Ozaki 1996**)**

Disrupted water quality

Contaminants
Association of fish assemblage structure and environmental variables: Compare fish assemblage composition to chemical and physical environment. (Waite and Carpenter 2000).

Reach-level processes

Riparian function
Remote sensing assessment: Use remote sensing classifications of vegetation to assess riparian buffer width and type in order to help determine how much and where riparian buffer impairment has occurred on a reach, watershed, and river basin scale. Identify historic conditions using reference

locations and historic documentation. Compare historic condition to current riparian condition in order to determine degree of change.

Field inventory: In addition to documenting forested buffer width, field inventories also classify stand types by species mix and seral stage, which gives sufficient information to prescribe generalized management regimes for each segment of riparian forest. Inventories also identify areas of livestock access and potential fencing projects.

Riparian alteration due to grazing: Use similar methods but include indicators such as stubble height measurements as indicator of disturbance (Clary and Lenninger 2000, Turner and Clary 2001).

Channel and floodplain interactions

Floodplain areas can be delineated using 100-year floodplain maps from the Federal Emergency Management Agency maps or U. S. Geological Survey 7.5-minute quadrangles and aerial photographs.

Habitat connectivity – anthropogenic blockages

Man-made barriers to anadromous fish habitat are identified through a systematic field inventory of channel crossing structures (culverts, tide gates, bridges, dams, and other manmade structures). An inventory identifies the type and physical dimensions of structures as well as physical attributes necessary for modeling water flow conditions and comparing results to passage criteria for salmonids (e.g., WDFW 1998).

Table 4-2. Summary of contemporary spatial scale classifications (adapted from Bauer and Ralph1999).

Classification system	Spatial scales addressed by classification system						
	Eco-region	River Basin	Watershed	Sub-watershed	Valley segment	Stream reach	Habitat unit
Bisson et al. 1982							X
Frissell et al. 1986			X	X	X	X	X
Seaber et al. 1987		X	X	X			
Paustian 1992				X	X	X	
Maxwell et al. 1995	X	X	X	X	X	X	
Rosgen 1996					X	X	
Montgomery and Buffington 1997						X	
Omernik and Bailey 1997	X	X	X				

Table 4-3. The 10 metrics of the benthic index of biological integrity (B-IBI), and their predicted response to increasing human disturbance.

Metric	Description	Response
Taxa richness and composition		
Total taxa	Richness	Decrease
Mayfly taxa	Richness	Decrease
Stonefly taxa	Richness	Decrease
Caddisfly taxa	Richness	Decrease
Population structure		
Dominance by top 3 taxa	Relative abundance	Increase
Long-lived taxa richness	Richness	Decrease
Tolerance and intolerance		
Intolerant taxa richness	Richness	Decrease
Tolerant taxa	Relative abundance	Increase
Feeding and other habits		
Clinger taxa richness	Richness	Decrease
Predators	Relative abundance	Decrease

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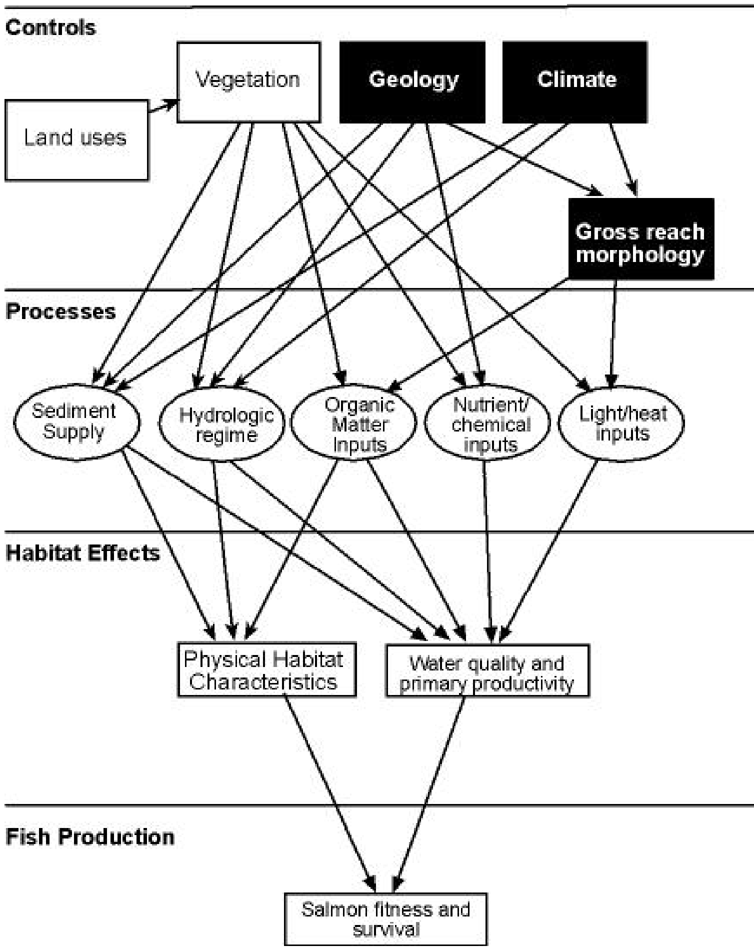

Figure 4-1. Schematic diagram of relationships between controls on watershed processes, effects on habitat conditions, and salmon survival and fitness (adapted from Beechie and Bolton 1999). Dark boxes in upper row are ultimate controls, light boxes are proximate controls.

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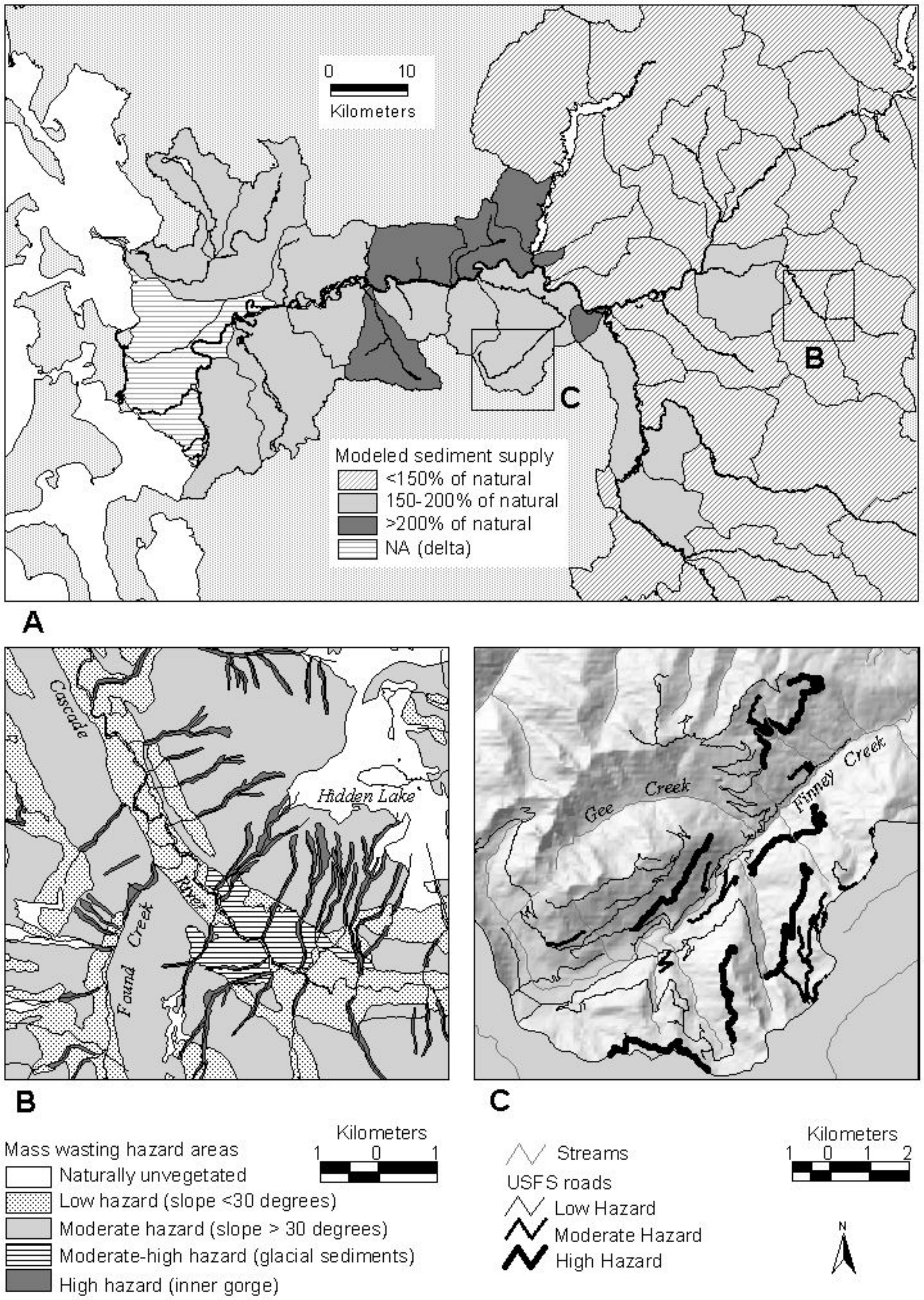

Figure 4-2. (A) Map of areas in the Skagit basin where sediment supply has likely increased due to land use, based on extrapolation of data from sediment budgets (described in text). (B) Landslide hazard map for a portion of the upper Cascade River basin. (C) Hazard map of U.S. Forest Service Roads classified as high risk of failure, moderate risk, or low risk.

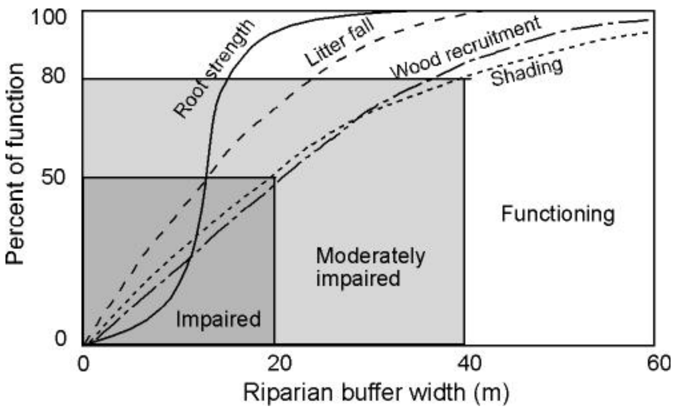

Figure 4-3. Illustration of change in riparian function with distance from channel (curves adapted from Sedell et al. 1997), and the Skagit Watershed Council's classification of impaired, moderately impaired, and functioning riparian forests.

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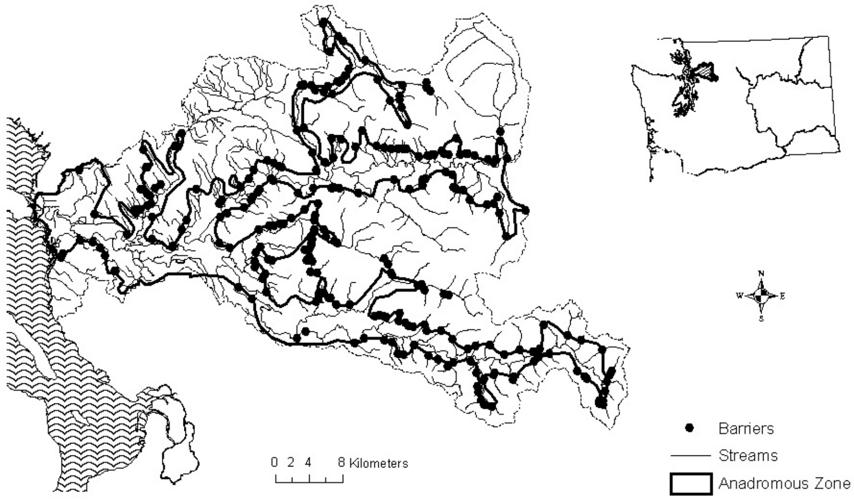

Figure 4-4. Example map of a portion of the inventoried stream crossing structures for a portion of the Stillaguamish River basin. The map shows structures that are blocking. Dark line indicates the historical extent of anadromous fish.

PRIORITIZING POTENTIAL RESTORATION ACTIONS WITHIN WATERSHEDS

Philip Roni, Timothy J. Beechie, and George R. Pess

Previous sections have outlined methods for assessing habitat loss and degradation, estimating fish response to changes in habitat, and developing a list of potential restoration actions. The methodologies outlined in these sections help identify disrupted processes and restoration opportunities. For example, a culvert inventory would provide a list of opportunities for culvert replacement to improve fish access, an inventory of riparian areas might identify opportunities for replanting, thinning, or fencing, or an inventory of diversions and water usage might indicate reaches in need of additional instream flows (see also the "Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions" section). Following these watershed assessments and identification of restoration opportunities, the next step is to prioritize specific restoration actions within a watershed (Beechie and Bolton 1999, OWEB 2001, Roni et al. 2002a). Initially, this requires an understanding of the watershed process or function that specific technique is likely to restore, the effectiveness and likelihood of success of each technique, and the potential fish response to that technique (Table 5-1; see also the "Introduction" and "An assessment Approach for Habitat Recovery Planning" sections).

Restoration actions may be prioritized based on a number of factors, including the needs of individual species, locations of refugia, or cost-effectiveness (Beechie and Bolton 1999). In this section we provide a brief overview of restoration, discuss different methods of prioritizing habitat restoration, and illustrate how different strategies can result in different project sequencing. We start with Roni et al. (2002a), which provides a general template for prioritizing restoration based on restoring watershed processes and known effectiveness of different techniques. We then outline other strategies for prioritizing restoration that consider cost, endangered species (refugia), potential fish production, and other factors that may influence salmon recovery. Finally, we discuss the need for long-term monitoring and treating restoration actions experiments.

What Is Restoration and What Do We Know about It?

The term "restoration" has been used to describe a suite of stream, watershed, and estuarine habitat manipulations, enhancements, and improvements. In its strictest definition, restoration is returning a site to some predisturbance condition (Gore 1985, NRC 1992). Restoration is generally considered different from habitat creation, reclamation, rehabilitation, and enhancement, in that it is more holistic or systemic and not accomplished through manipulation of individual ecosystem or watershed elements (NRC 1992; Frissell and Ralph 1998). Rather, habitat enhancement is the improvement of habitat from it's existing or previous condition and does not necessarily seek to restore conditions to some predisturbance state or seek to restore disrupted watershed or ecosystem processes and functions such as delivery of water, wood (organic material) and sediment. Restoration can also be further classified as passive and active (Kauffman et a. 1997). Passive techniques seek to restore processes by halting detrimental land uses, protecting areas and setting up conditions that will allow recovery of the stream (e.g., exclusion of cattle from riparian areas, protecting riparian area). Active restoration generally seeks to create relatively rapid habitat changes (within a few months or years) and treat symptoms of disrupted watershed processes rather than restore the process. Active techniques, which may include habitat enhancement, are those that seek to directly improve or manipulate habitat, such as removal of migration barrier or placement of logs in a stream channel to create pools, or thinning of riparian areas. Here we will use the term restoration generically to mean both restoration and enhancement, but we will distinguish between those activities that restore watershed or ecosystem processes and those that enhance habitat (Figure 5-1).

Before one can prioritize specific restoration actions within a watershed, a comprehensive understanding of the physical and biological effectiveness of various restoration methods is needed. Numerous techniques exist to restore disrupted processes or enhance habitat in the short term. Most restoration techniques fall into five general categories: 1) habitat reconnection, 2) road improvement, 3) riparian restoration, 4) instream habitat restoration, and 5) nutrient enrichment. Within each of these five general categories, several specific techniques can be identified. For example, several types of wood placement have been developed to increase instream habitat complexity and loss of large woody debris (LWD).

Gore (1985), Reeves et al. (1991), Slaney and Slodakas (1997), Cowx and Welcome (1998), OWEB (1999b) and others provide descriptions of various restoration techniques, and information on design and implementation of these techniques. Several authors have reviewed the effectiveness of different restoration techniques, though rather limited information exists for most techniques (Gore 1985, Reeves et al. 1991, Roni et al. 2002a). Roni et al. (2002a) recently reviewed common restoration techniques, their effectiveness, longevity and whether they restore processes or are short-term habitat enhancement (Table 5-1). However, there is rather limited specific guidance on which techniques to use and how to prioritize restoration, though many authors have discussed the need to prioritize restoration and restore processes (Beechie et al. 1996, Minns 1996, Jones and Moore 2000, Rieman et al. 2000, Luce et al. 2000, JNRC 2002).

In part, the lack of guidance on the appropriateness of various techniques and how to prioritize restoration actions stems from limited information on the biological effectiveness of various habitat restoration and enhancement techniques (Reeves et al. 1991, Chapman 1996, Roni et al. 2002a, 2002b). The responses of fishes to watershed and stream habitat restoration techniques have not been thoroughly evaluated, and there is considerable debate within the scientific community about the effectiveness of various techniques (Reeves et al. 1991, Kondolf 1995, Kauffman et al. 1997, Roni et al. 2002a).

Most monitoring has focused on the physical response to various instream restoration techniques with inadequate monitoring of fish, invertebrates, and other biota. Response of fish and other biota are inherently more difficult to monitor than physical conditions. However, the biological response to various restoration techniques is often the ultimate measure of effectiveness. The large interannual variability of juvenile and adult salmonid abundance often requires more than 10 years of monitoring to detect a response to restoration (Bisson et al. 1995, Reeves et al. 1997, Maxell 1999, Ham and Pearsons 2000, Roni et al. 2002b). Existing monitoring has also indicated highly variable results from some techniques such as wood and boulder placement in streams (Chapman 1996). Therefore, drawing conclusions about the biological effectiveness of various techniques has been difficult and has hampered efforts to provide scientific guidance on restoration activities.

In the 1990s it became widely accepted that restoring watershed processes is the key to restoring watershed health and improving fish habitat throughout western North America and elsewhere. Beechie et al. (1996), Kauffman et al. (1997), Beechie and Bolton (1999), Roni et al. (2002a) and others have described restoration and recovery strategies that place emphasis on restoring physical and biological processes that create healthy watersheds and high-quality habitats. Yet activities that restore processes (e.g., road removal and restoration, culvert removal, and riparian and upslope restoration) are often conducted at the site or reach level. Prioritization of restoration actions needs to place site-specific restoration within a watershed context.

Strategies for Prioritizing Actions

Watershed and stream restoration are a key component of many land management plans, and should be an important component of most recovery plans for threatened and endangered species. It is often unclear how individual site-specific actions might fit into a larger context of watershed restoration and recovery of salmon stocks. Describing a single prioritization scheme that will be applicable to all watersheds is difficult; therefore, we first describe a general strategy for prioritizing site-specific restoration activities in a watershed context and then discuss other methods and factors to consider in prioritizing restoration. We focus on those activities that occur within an individual watershed (U.S.G.S. 5 or 6th field hydrologic units - HUCs). Other large-scale restoration efforts that may occur at the basin or ESU scale, and thus may influence many watersheds, should be prioritized within a basin or ESU. We do not discuss such broad-scale priorities here.

In prioritizing restoration, it is important to consider the response time, probability and variability of success, and the duration of a given restoration action (Table 5-1). All else being equal (e.g., costs, listed species concerns), those techniques that have a high probability of success, low variability among projects, and relatively quick response time should be implemented before other techniques. For example, reconnecting isolated off-channel habitats or blocked tributaries provides a quick biological response, is likely to last many decades and, based on available evidence, has a high likelihood of success. Generally, these types of restoration activities should be undertaken before methods that produce less consistent results. Riparian restoration or road improvement may not produce results for many years or even decades for some functions (Table 5-1) and should be considered after reconnecting high-quality isolated habitats. Other techniques, such as instream LWD placement or other instream restoration, are generally effective at increasing coho salmon densities, but less certain for other species. Instream actions such as LWD placement are often habitat manipulations or enhancements and should be undertaken either after or in conjunction with reconnection of isolated habitats and other efforts to restore watershed processes. In addition, manipulation of instream habitat may be appropriate where short-term increases in fish production are needed for a threatened or endangered species (Beechie and Bolton 1999).

Using information summarized in table 5-1, Roni et al. (2002a) developed a hierarchical flow chart that can be used to help guide the selection and prioritization of restoration projects based on understanding of watershed processes and current knowledge on effectiveness of different techniques (Figure 5-2). This flow chart combines the known effectiveness of various techniques with the need to restore habitat-forming processes (Figure 5-1) identified by watershed assessments (the “Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions” section) and the protection of high-quality habitats. Protection of high-quality habitat should be given priority over habitat restoration, as it is far easier and more successful to maintain good habitat than to try and recreate or restore degraded habitat.

While most techniques fit well into this hierarchy, estuarine restoration, carcass placement and nutrient enhancement are relatively new techniques whose place in this hierarchy is uncertain. Little is known about the effectiveness of estuarine restoration, though reconnecting isolated estuarine habitats such as distributary sloughs is similar to reconnecting isolated off-channel habitats, which has been shown to be effective (Table 5-1). Given the importance of estuaries to anadromous fishes and the success of reconnecting isolated off-channel habitats, it is likely that reconnecting estuarine habitat would be effective and should be considered at the same time as reconnecting other isolated habitats. The placement of salmon carcasses or other nutrients into streams may increase fish condition and production in the short term. This restoration technique is a form of habitat enhancement that can occur at any stage in the watershed restoration process. However, because it does not restore but rather mitigates for a deficient process, we have suggested that it be considered at the same point in the hierarchy as instream habitat manipulation. Similarly, the creation of new estuarine or off-channel habitats does not restore a process and the effectiveness of these efforts is unclear.

A common restoration technique not covered in Roni et al. (2002a) is restoration of instream flows or natural hydrology either from water withdrawal projects or below large water storage projects. Water withdrawal or flow manipulation disrupts hydrologic processes, including delivery and routing of sediment and nutrients, and can dramatically impact habitat formation, connectivity, and quality (Bednarek 2001). We consider restoring instream flows and natural hydrologic patterns part of reconnecting isolated habitats, and therefore do not have a separate category for this technique. Upslope activities and land use can have dramatic effects on stream hydrology, sediment delivery, water chemistry, and water quality. Altering land use and other upslope restoration techniques were also not explicitly discussed in Roni et al. (2002a) either, but essentially can be included with riparian restoration (Table 5-1).

Within the broad restoration categories in Figure 5-2, some techniques are more effective than others or more applicable in some provinces than others. For example, we include riparian silviculture with fencing and reduced grazing under riparian restoration. Livestock exclusion is a form of riparian protection that has been shown to be effective on range and agricultural lands (Platts 1991), while the long-term effectiveness of riparian replanting and conversion in forested watersheds techniques is largely unknown. Priorities for different types of riparian restoration will differ by region and watershed, as will other specific restoration techniques that fall into the broad categories we have defined. However, a watershed assessment is the important first step to determine the most effective type of restoration within a given restoration category for the watershed in question (Beechie et al. in press).

The principles outlined above and in Figure 5-1 were designed primarily for forest, range, and other moderately modified rural lands. However, they are still useful in urban and agricultural lands, even though other factors such as large infrastructure (e.g., highways and buildings) may constrain certain restoration opportunities. In urban areas, hydrologic and sediment processes in streams are highly altered (e.g., increased high flows and channel down-cutting). Areas with intensive agriculture often have severe water quality problems, and stream channels in both urban and agricultural areas are often highly channelized and lack adequate riparian vegetation. Thus the framework we outline may need to be modified for use in these highly altered systems where some processes cannot be reliably restored, or where water quality or hydrologic changes may compromise the effectiveness of many of the commonly employed restoration techniques.

Alternative Prioritization Schemes

Alternative strategies for prioritizing restoration that incorporate economics, biologic, ecologic, and biologic factors have been proposed or used. For, example Beechie and Bolton

(1999) provide an example of how the sequencing of habitat restoration might change based on species interest (Figure 5-3). Sedell et al. (1995), Wasserman et al. (1995), Beechie et al. (1996), Frissell (no date), Frissell and Bayles (1996), and others have outlined restoration strategies that focus on providing refugia and protecting high quality habitats. Beechie et al. (1996) outlined a prioritization strategy that focused on providing refugia for a depressed steelhead stock in Deer Creek, Washington. Other strategies might prioritize actions on potential increase in fish numbers, cost, cost per fish, aquatic diversity, assuring for metapopulations structure or diversity, or scoring based on a suite of these and other factors (e.g., Beechie et al. 1996, Frisell and Bayles 1996, Doyle 1997, SRSRC 2002, LCFRB 2002). Some states, such as Oregon, have developed sequential methodologies for conducting assessments, prioritizing, and implementing restoration activities (Figure 5-4). These various strategies incorporate management goals beyond simply restoring watershed or ecosystem processes and habitat. Thus the sequencing of restoration actions under different prioritization strategies will vary.

We demonstrate how priorities might differ based on different restoration prioritization schemes by running alternative scenarios. First, we developed a hypothetical list of potential restoration actions along with detailed information on their cost, length and area restored, whether they provide refugia for endangered species, potential increase in fish numbers, and cost per fish (Table 5-2). Second, we ranked restoration actions using different prioritization schemes mentioned above (Table 5-3). This analysis demonstrated that if restoration actions were prioritized based on Roni et al. (2002a), impassible culverts and reconnection of habitats would occur first, followed by road, riparian, and LWD placement. If actions were prioritized by whether they were in a refugia for an ESA listed species, instream flow and LWD placement would be first, simply because they are in a high priority area. Similarly, different cost, cost/fish, total fish production all produced slightly different prioritization scenarios. This simple example illustrates how priorities might differ based on the method, information used, and management objectives. In the section on "Managing Uncertainty in Salmon Habitat Recovery Planning," we also discuss accounting for uncertainty in prioritizing restoration actions toward incorporating the risks and likelihoods of success and failure (physical, biological, or financial) into the planning process.

The appropriate method for prioritizing restoration activities within a watershed will depend on numerous factors. Our intent in this section has been to discuss how one prioritizes site-specific restoration actions within a watershed or basin. However, all else being equal or if limited information is available, we would recommend a strategy similar to that outlined in Roni et al. (2002a; Figure 5-2) that focuses on reconnecting isolated habitats and restoring watershed processes before or alongside habitat manipulations or enhancement.

Need for Monitoring and Management Experiments

Reviews of various restoration techniques (e.g., Roni et al. 2002a) indicate that knowledge about the effectiveness of most techniques is incomplete, and comprehensive research and monitoring are needed. Even techniques that appear to be well studied, such as instream LWD placement, need more thorough evaluation and long-term monitoring. This emphasizes the need for comprehensive monitoring and evaluation of both individual and multiple restoration actions at multiple scales. Many restoration actions should be treated as management experiments and accompanied by research and monitoring to determine both physical and biological responses. These results can then be used to guide future restoration actions and more accurately quantify the potential increase in fish production for habitat manipulations. Ultimately, thorough monitoring and evaluation of restoration actions will help us prioritize restoration opportunities and wisely spend limited restoration and recovery funds for salmon.

Table 5-1. Typical response time, duration, variability in success and probability of success of common restoration techniques (modified from Roni et al. 2002a). The first three categories of restoration (reconnect isolated habitats, roads and land use, and riparian restoration) are considered process-based or passive restoration, the last three (instream, nutrient enrichment, and habitat creation) are considered enhancement or active restoration.

Restoration type	Specific action	Years to achieve response	Longevity of action (y)	Variability of success among projects	Probability of success
Reconnect habitats	Culverts	1-5	10-50+	Low	High
	Off channel	1-5	10-50+	Low	High
	Estuarine	5-20	10-50+	Moderate	Moderate to high
	Instream flows	1-5	10-50+	Low	High
Roads and land use	Road removal	5-20	Decades to centuries	Low	High
	Road alteration	5-20	Decades to centuries	Moderate	Moderate to high
	Change in land use	10+	Decades to centuries	Unknown	Unknown
Riparian restoration	Fencing	5-20	10-50+	Low	Moderate to high
	Riparian replanting	5-20	10-50+	Low	Moderate to high
	Rest-rotation or grazing strategy	5-20	10-50+	Moderate	Moderate
	Conifer conversion	10-100	Centuries	High	Low to moderate
Instream habitat restoration	Artificial log structures	1-5	5-20	High	Low to high[a]
	Natural LWD placement	1-5	5-20	High	Low to high[a]
	Artificial log jams	1-5	10-50+	Moderate	Low to high[a]
	Boulder placement	1-5	5-20	Moderate	Low to high[a]
	Gabions	1-5	10	Moderate	Low to high[a]
Nutrient enhancement	Carcass placement	1-5	Unknown	Low	Moderate to high
	Stream fertilization	1-5	Unknown	Moderate	Moderate to high

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Habitat creation	Off channel	1-5	10-50+	High	Moderate
	Estuarine	5-10	10-50+	High	Low
	Instream	See various instream restoration techniques above			

[a] Depends on species and project design

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Table 5-2. Example of list of potential restoration actions within a watershed. Refugia are based on Beechie et al. (1996) and include 1= refugia (areas where recovery is relatively predictable), 2 = key habitat areas or ears provide for the largest long –term recovery of species of interest, but are sensitive to disturbance and more difficult to restore, and 3 = key habitat areas or areas expected to provide the smallest gain for species of interest. Coho and chinook smolts represent expected annual increase in smolt production. All numbers and stream names are fictitious and for demonstration purposes only.

Site ID	Site name	Action	Refu-gia	Km treat-ed	M^2 treated	Coho smolts	Chinook smolts	Cost	$ per coho
A	Clark Creek	LWD placement	2	3	15,000	3,750	750	50,000	13.30
B	Upper Simpson Creek	LWD placement	1	2	10,000	2,500	500	32,000	12.80
C	Lower Simpson Creek	LWD placement	1	2	14,000	3,500	700	35,000	10.00
D	Check Creek	Fencing	1	5	60,000	6,000	3,000	20,000	3.30
E	Dry Creek	Increase flows	2	20	200,000	20,000	10,000	500,000	25.00
F	Big River	Reconnect tidal channels	3	1	100,000	10,000	50,000	350,000	35.00
G	Big River	Create new estuarine slough	3	2	200,000	20,000	100,000	750,000	37.50
H	Clark Creek	Culvert replacement/ fish passage	2	3	15,000	3,750	0	150,000	40.00
I	Simpson Creek	Road de-commissioning	1	20	200,000	20,000	10,000	1,500,000	75.00
J	Clark Creek	Road resurfacing/ sediment reduction	2	10	50,000	5,000	2500	750,000	150.00
K	Big River Slough	Reconnect isolated oxbow slough	3	4	800,000	400,000	40,000	75,000	0.19

Table 5-3. Example of different order of priorities based on use of different prioritization schemes using information presented in Table 5-2. Roni et al. (2002a) and refugia methods of prioritization do not distinguish between projects of the same type. Hence, there are only four levels and three levels, for the two methods, respectively.

Site ID	Potential Restoration Action	Roni	Refugia	Total Cost	Cost/Coho	Total Fish
A	LWD placement	3	2	4	5	4
B	LWD placement	3	1	2	4	1
C	LWD placement	3	1	3	3	3
D	Fencing/cattle exclusion	3	1	1	2	6
E	Instream flows/purchase water rights	1	2	8	6	8
F	Reconnect estuarine tidal channel (dike removal)	1	3	7	7	9
G	Excavate new estuarine slough	4	3	10	8	10
H	Culvert replacement/fish passage	1	2	6	9	2
I	Road decommissioning	2	1	11	10	7
J	Road resurfacing/sediment reduction	2	2	9	11	5
K	Reconnect isolated oxbow slough	1	3	5	1	11

FOR REVIEW ONLY

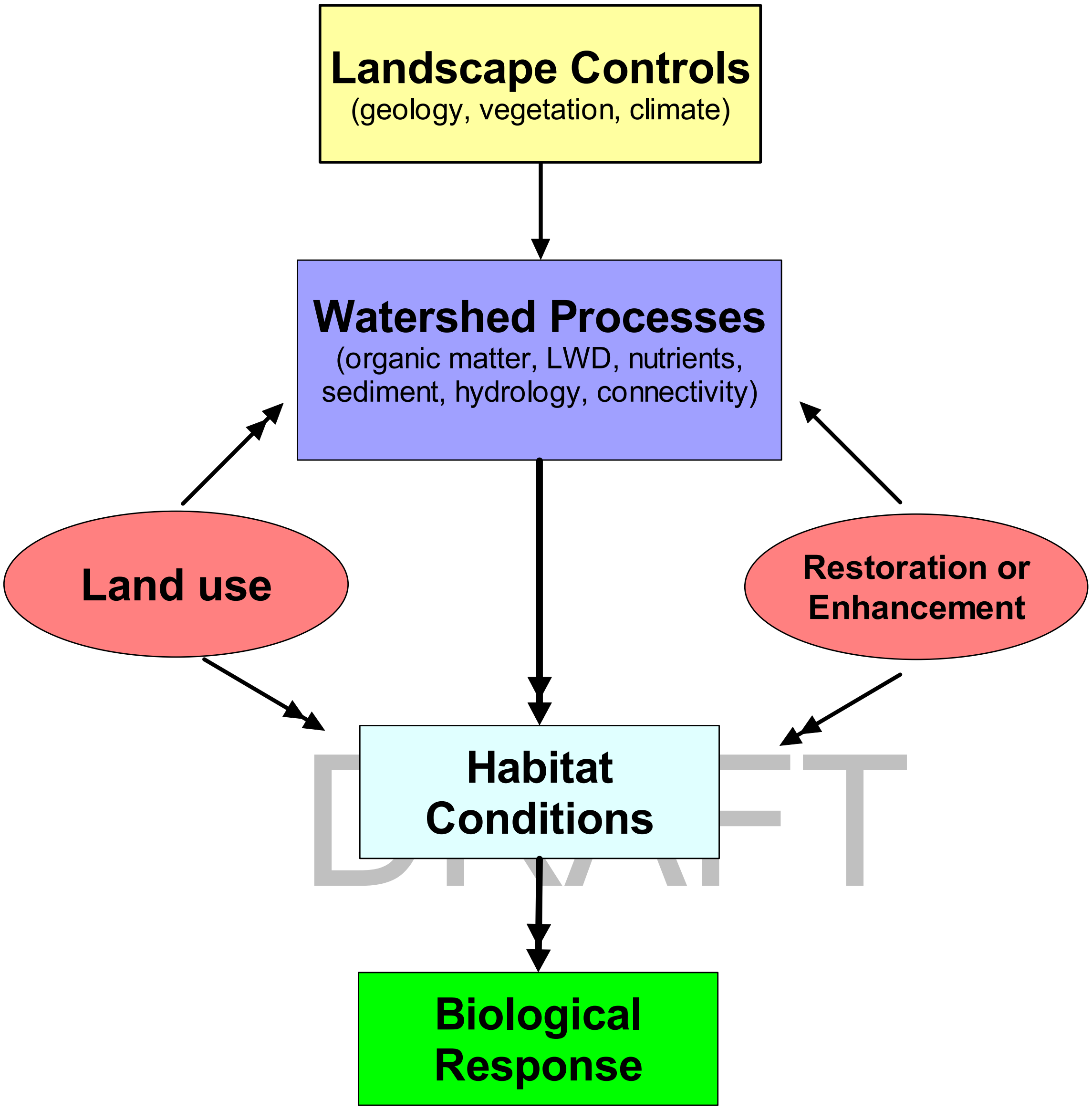

Figure 5-1. Simplified model of watershed controls, processes, and function and how land use, restoration and enhancement can influence habitat and biota.

FOR REVIEW ONLY

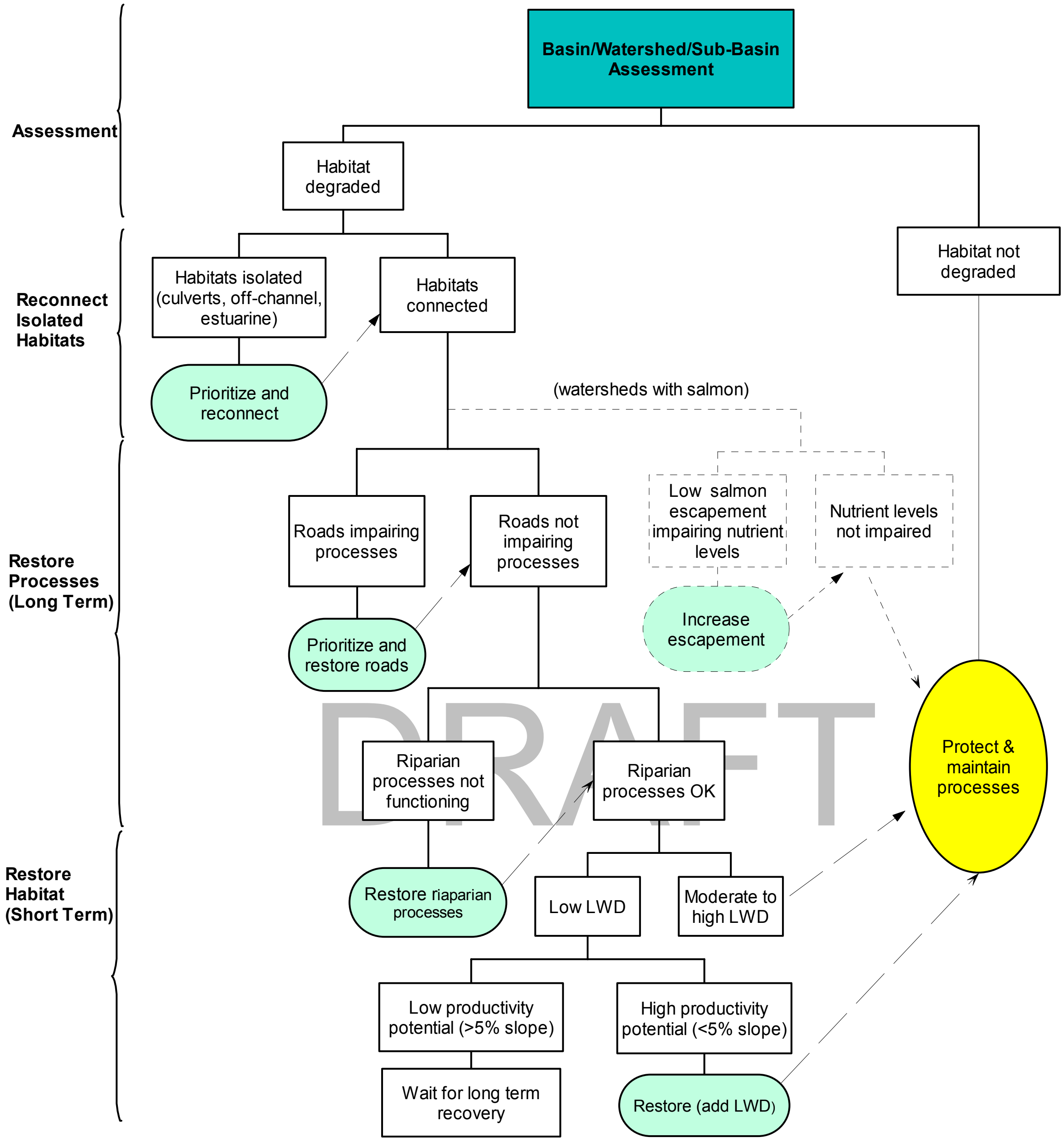

Figure 5-2. Flow chart depicting hierarchical strategy for prioritizing specific restoration activities (modified from Roni et al. 2002a). Shaded boxes indicate where restoration actions should take place. Addition of salmon carcasses or nutrients may be appropriate at various stages following reconnection of isolated habitats.

FOR REVIEW ONLY

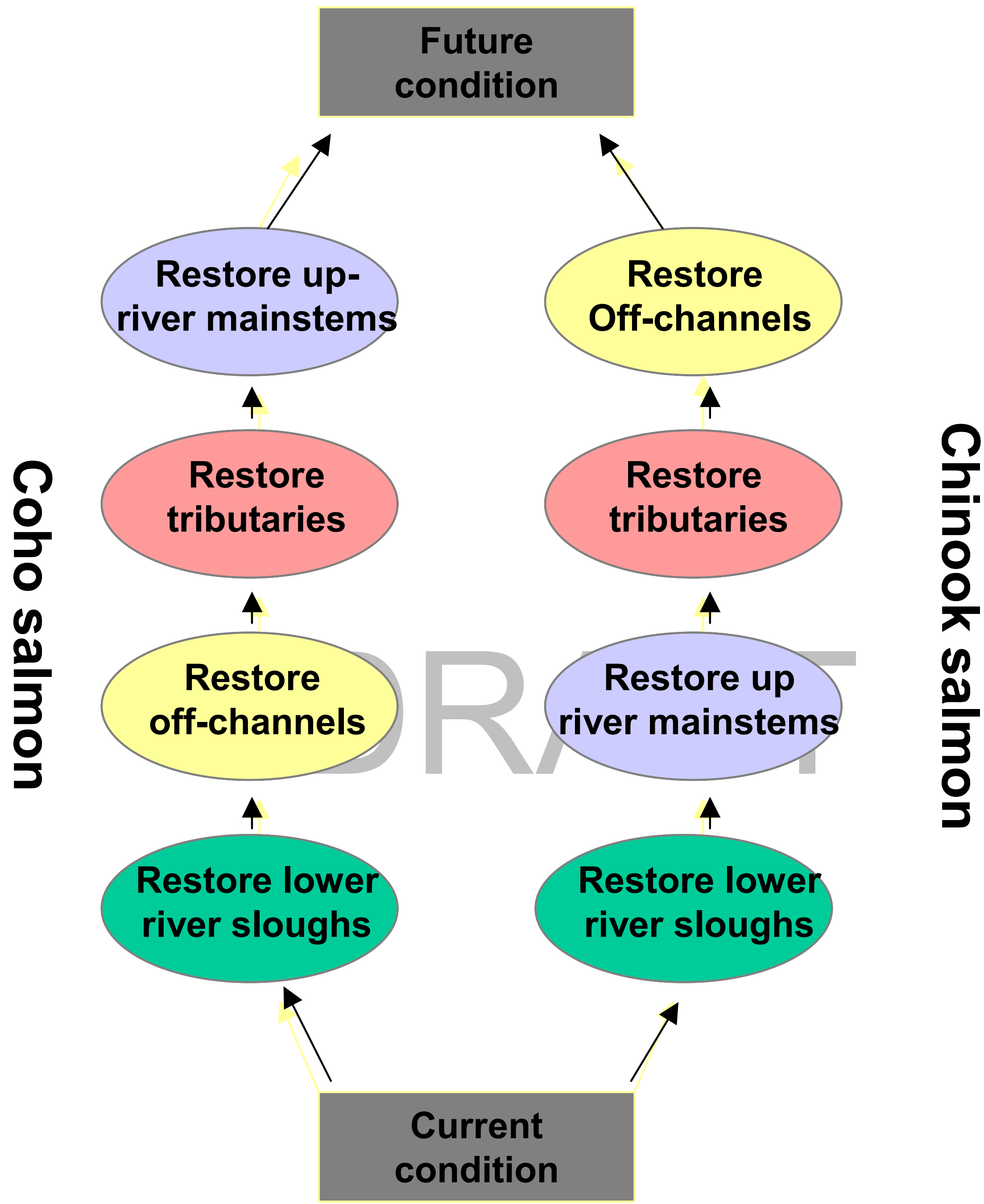

Figure 5-3. Sequence and prioritization of habitat restoration based on species of interest. Modified from Beechie and Bolton (1999).

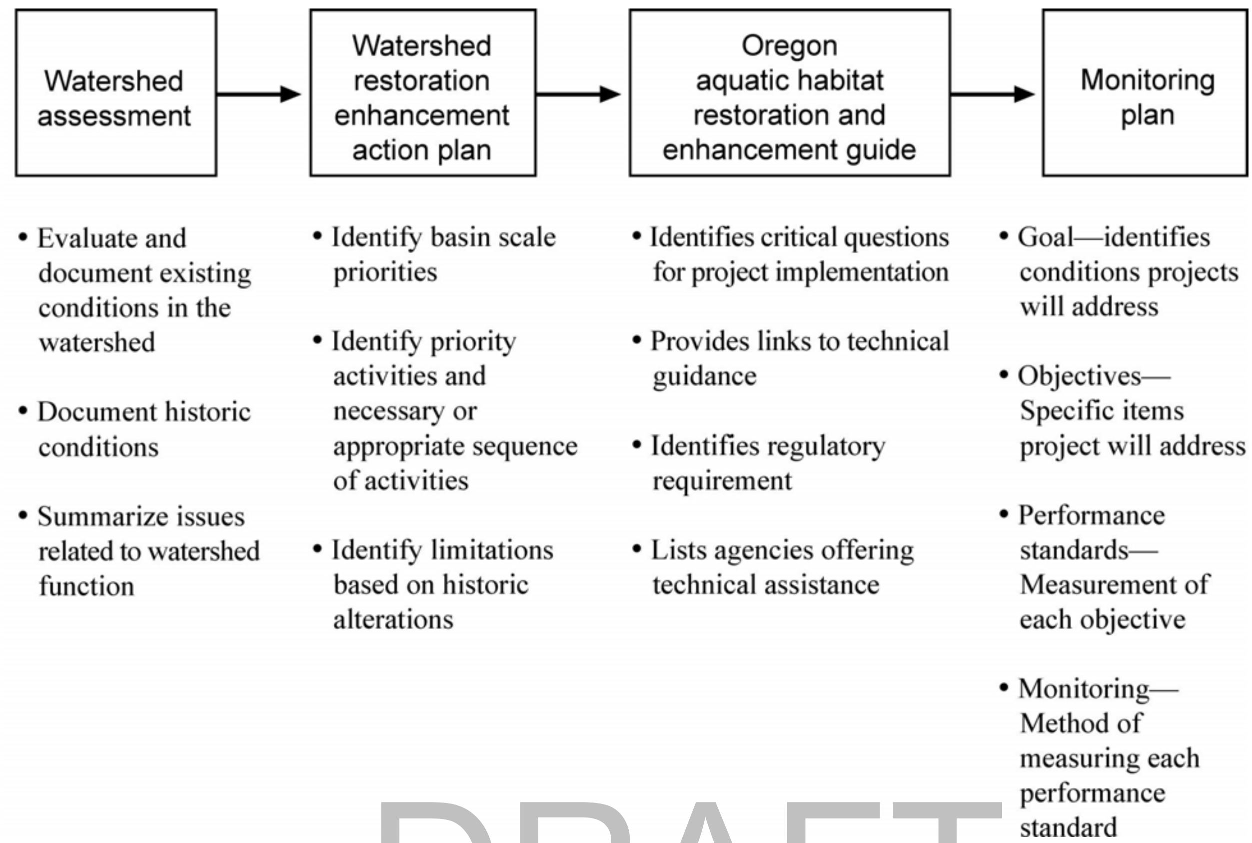

Figure 5-4. Process for restoration planning, prioritization, and implementation used by the Oregon Watershed Enhancement Board, Oregon Department of Fish and Wildlife, and state of Oregon. Figure reproduced from Oregon Aquatic Habitat Restoration and Enhancement Guide (OWEB 1999b).

ISSUES OF SCALE IN HABITAT RECOVERY PLANNING

Cara A. Campbell

The aquatic environment is complex and dynamic, changing continually across space and time. Inhabitants of this environment have evolved in response to these ever-changing conditions. However, anthropogenic alterations to the landscape have disrupted the natural processes within these systems and species are often forced to contend with altered or unnatural habitat conditions. These alterations can be large or small, influencing expansive areas or more local conditions, and the effects can occur immediately or years later. Thus, a thorough knowledge of the processes structuring the aquatic environment and how these processes interact over various spatial and temporal scales is critical for understanding the effects of disturbance on aquatic systems and their inhabitants.

This section will examine the concept of scale and how it can be incorporated into recovery planning, with a particular emphasis on analyses to help set recovery goals. Similar concepts apply to analyses designed to identify ecosystem recovery actions, which are discussed in the "Prioritizing Potential Restoration Actions Within Watersheds" section on page XX and the "Updating the Recovery Plan" subsection on page XX. First, the inherent hierarchical nature of aquatic systems will be discussed, specifying the need to study the processes and inhabitants of these systems in a similar hierarchical manner. A review of small- and large-scale studies will follow, focusing on the difficulties in transferring information across scales. Finally, these concepts will be incorporated into examinations of how habitat alterations might influence the four types of recovery goals: abundance, productivity, diversity, and spatial structure (McElhany et al. 2000). Examples from the literature will be used to illustrate what information can be obtained at each scale as well as to show how combining studies across scales can provide information on both the processes impaired as well as on recovery potential.

Hierarchical Nature of Stream Systems

The aquatic environment can be viewed as a hierarchy of spatially nested systems (Frissell et al. 1986, Urban et al. 1987). Implicit in this hierarchical model is the concept of scale. Each of these spatially nested systems can be thought of as an individual spatial scale (Figure 6-1). For example, a reach (10^1 m) occurs at a smaller spatial scale than the watershed (10^3 m) within which it is contained. Similarly, the evolutionary and developmental processes within each level occur over different time frames, i.e. over different temporal scales (Figure 6-1). For example, infrequent (10^6–10^5 years), high magnitude geologic events cause evolution of watersheds, while frequent (10^1-10^0 years), low magnitude geomorphic events cause evolution of smaller, habitat units (Frissell et al. 1986). The result is a system in which development and persistence occur at specific temporal scales within each level of the hierarchy such that conditions within smaller-scale systems are constrained by the larger-scale systems within which they are contained (Frissell et al. 1986, Urban et al. 1987).

In addition, longitudinal (upstream to downstream) and lateral (stream to terrestrial) linkages help shape biological and physical structure at each level resulting in a predictable spatiotemporal gradient of physical and biological conditions from headwaters to mouth (Vannote et al. 1980, Frissell et al. 1986, Gregory et al. 1991). For example, stream width, depth, discharge, (Platts 1979, Leopold et al. 1964), temperature (Allan 1995), and biological diversity (Vannote et al. 1980, Barila et al. 1981) increase with stream size while gradient and substrate size decreases (Platts 1979, Leopold et al. 1964). Also, a decrease in terrestrial inputs and riparian shading coupled with an increase in organic transport occurs from headwaters to estuary (Vannote et al. 1980). Moreover, these conditions are structured by the climate, geology, and anthropogenic activity of the specific watershed (Vannote et al. 1980, Frissell et al. 1986). As a result, each segment within the system contains a predictable array of habitat conditions dependent on the watershed; however, these habitats are not homogenous -- there is simply an order to their heterogeneity (Frissell et al. 1986).

This habitat heterogeneity implies that species distributions will not be even across the landscape but will instead occur in patches, varying with the quality and quantity of specific habitats. The quality and quantity of habitat is a product of both large- and small-scale processes, and it is necessary to quantify both the significance of this environmental variability as well as the subsequent biotic responses (Cunjak 1996). For example, water temperature varies both spatially and temporally and influences salmonid life history accordingly. Large-scale temperature regimes dictate species ranges as well as species distributions (Meisner 1990, Flebbe 1993, Welsh et al. 1995, Keleher and Rahel 1996, Welsh et al. 1998). At the watershed scale, elevation, latitude, aspect, and stream size interact to determine annual and seasonal temperature cycles. Thus, the physical location of a stream within the river network influences population life-history characteristics such as spawn timing (Gresswell et al. 1997), growth rate (Lobón-Cerviá and Rincón 1998, Campbell 1999), and the timing of smolt migration (Whalen et al. 1999*a*). At smaller reach or segment scales, the type and density of riparian vegetation and the degree of groundwater input influences the stability of seasonal and diel temperatures (Smith and Lavis 1975, Gregory et al. 1991, Allan 1995) which, in turn, can influence individual holding performance (Rimmer et al. 1985, Graham et al. 1986), behavior (Fraser et al. 1993), food digestion and assimilation (Cunjak and Power 1987, Cunjak et al. 1987), and inter- and intra-specific survival and distribution (Campbell 1999, Torgersen et al. 1999, Harvey et al. 2002). Finally, at the habitat unit scale, cool water inputs from tributaries, intergravel flow through river bars, and streamside subsurface sources can thermally stratify individual pools, providing cool water refuges for individuals of multiple age-classes (Nielsen et al. 1994). Thus, species assemblages and distributions are structured by a combination of larger-scale geomorphic and climatic conditions and the more specific biotic and abiotic conditions of the local environment. As a result, the biological and physical conditions at any site must be examined in the context of the larger geologic, climatic, and geomorphologic conditions of the system as a whole -- a multi-scale approach.

Transferability Across Scales

Smaller-scale studies generally focus on identifying physical features used by individuals (Bustard and Narver 1975, Cunjak 1988, Nakamoto 1994), how this habitat use changes ontogenetically and temporally (Rimmer et al. 1983, Rimmer et al. 1984, Baltz et al. 1991, Heggenes et al. 1993, Whalen et al. 1999b), and how these habitat preferences differ by species (Bisson et al. 1988, Fausch 1993, Heggenes et al. 2002). These types of studies have also identified bottlenecks limiting the production of different salmonids (McMahon and Hartman 1989, Tschaplinski and Hartman 1983, Solazzi et al. 2000), and have contributed to our understanding of both intra- (Symons and Heland 1978, Kennedy and Strange 1986, Harvey and Nakamoto 1997) and inter-specific (Hearn and Kynard 1986) competition. These smaller-scale studies often generate models based on correlations between habitat use and/or availability and fish abundance, incorporating the concepts of optimal and suitable habitat. Problems can arise because individuals are not always found in suitable habitat (Bozek and Rahel 1991) and not all recruits are necessarily produced from those habitats with the highest densities (Grossman et al. 1995). However, some preferences (e.g., nose velocities) are transferable within systems (DeGraaf and Bain 1986, Morantz et al. 1987), while for other habitat characteristics (e.g., substrate and depth) occupancy varies with availability across systems (Bozek and Rahel 1992). The low predictive power of these habitat-based models may result from: 1) the exclusion of biological factors such as abundances of prey, predators, or competitors (Grossman et al. 1995); 2) a failure to identify the specific mechanism(s) responsible for the selection, and the subsequent consequences of this habitat selection on individual fitness (Grossman et al. 1995); and 3) the failure to incorporate stream level variability (Dunham and Vinyard 1997). Also, stochastic disturbances common over smaller scales cause the behavior of these systems to be more variable, and thus, less predictable (Levin 1992). While, site-specific models might be the ideal for predictive purposes, collecting site-specific data is time consuming and expensive. Other methodologies must be developed to create models able to predict population characteristics across landscapes.

Large-scale investigations generally address questions regarding the role of climate (Meisner 1990, Keleher and Rahel 1996), geology and geomorphology (Platts 1979, Lanka et al. 1987, Nelson et al. 1992, Richards et al. 1996, Kruse et al. 1997), and land use and land cover (Connolly and Hall 1999, Bradford and Irvine 2000, Waite and Carpenter 2000, Paulsen and Fisher 2001) in shaping both aquatic systems and their inhabitants. Examining larger spatial scales over longer time frames produces more generalized models and subsequently offers greater predictability; however, detail is sacrificed (Levin 1992). Models link species presence, absence, or composition to broad characteristics within the stream hierarchy (e.g., stream size, geology, climate, land use) to create predictive correlations and, similar to smaller-scale models, show mixed results. The omission of natural temporal variation in population abundance can hamper results (House 1995, Bradford et al. 1997) as can study designs that are either not sufficiently spatially diverse (Baxter et al. 1999, Rieman and McIntyre 1996, Pess et al. 2002) or temporally long (Rieman and Myers 1997) to be reliably indicative of population trends and any subsequent associations with habitat or habitat change. Also, many larger-scale analyses synthesize existing small-scale datasets created for other purposes. This variability can sometimes result in datasets exhibiting inconsistent sampling frequencies, efficiencies, and representativeness, the analysis of which can produce unreliable, inexplicable results (Rieman et al. 1999). However, it is the combination of small-scale biotic and abiotic conditions experienced by an individual, within the constraints set by the larger landscape, which ultimately determines occupancy of a given habitat. As a result, there has recently been a rise in multi-scale investigations that seek to understand the larger-scale variables structuring aquatic species and, within these larger variables, the specific habitat characteristic influencing populations (e.g. Watson and Hillman 1997, Baxter and Hauer 2000, Labbe and Fausch 2000, Pess et al. 2002).

In sum, key information can be obtained at any scale of study; however, combining this knowledge across scales and disciplines has proven inherently difficult (Levin 1992, Imhof et al. 1996). As a result, there is increasing emphasis on maintaining a high degree of habitat heterogeneity to ensure the availability of sufficient habitat combinations to sustain multiple populations and species (Ward 1998). With this goal in mind, the focus should be on maintaining proper ecosystem functioning rather than on managing for specific habitat criteria. This focus requires a thorough understanding of the linkages between biological and physical processes within and across scales to effectively manage habitat and define recovery potential (Lewis et al. 1996).

Scale in Recovery Planning

Anthropogenic activities can have serious implications for population viability and persistence by affecting the quantity or quality of habitat. Forestry practices can alter the volume and timing of runoff as well as the sediment supply process, and can reduce the volume of large woody debris, the number of off-channel habitats, streambank stability, channel roughness, and water quality (Meehan 1991). Aquatic systems adjoining agricultural lands can be associated with reduced riparian vegetation, streambank instability, increased sedimentation, hydromodification, and higher levels of nutrients and pesticides (Waters 1995, Waite and Carpenter 2000). Urbanized areas experience increased sedimentation and pollution, along with many of the problems associated with agriculture (Waters 1995, Waite and Carpenter 2000). Dams and other forms of hydromodification can reduce or eliminate the natural flow regime, isolating river channels from its floodplain and riparian systems, and altering the natural processes of sediment erosion and deposition (Poff et al. 1997).

All these effects can alter salmonid abundances at various life-stages, as well as influence the maintenance and recovery of populations and Evolutionary Significant Units (ESUs). For example, species with rigid habitat requirements are under a stronger threat from degraded habitat conditions than those with flexible habitat requirements. Also, a species' degree of tolerance to degraded conditions can determine the likelihood of its displacement by another species (Nelson et al. 1992). Since greater habitat complexity is associated with greater community complexity (Gorman and Karr 1978), and since anthropogenic actions generally serve to reduce complexity (Ward 1998), understanding how the various land use activities alter natural processes and conditions within stream systems is critical for recovery efforts.

Successful restoration of both natural processes and salmon populations requires the incorporation of multi-scale information into the recovery planning process. The first step is to inventory and collect all available data, covering multiple temporal and spatial scales, across the area of consideration. This would initially focus on current and historical data of three types: 1) status of habitat forming processes and biological integrity, 2) condition and distribution of aquatic habitats, and 3) abundance and distribution of salmon. This information then needs to be examined to assess how habitat factors prevent populations or their parent ESUs from meeting the four types of recovery goals: abundance, productivity, diversity, and spatial structure (McElhany et al. 2000). The following subsections will discuss how scale can be incorporated into these assessments.

Abundance

The risks to a population are inversely related with abundance, so it should be possible to use abundance to define broad risk categories (McElhany et al. 2000). To address how habitat changes might have altered abundance and thus population risk, it is necessary to determine how abundance changes with land use. First, it is necessary to look for trends in population abundance through the use of population estimates such as redd counts, dam and weir counts, spawner and carcass surveys, harvest estimates, and juvenile counts. Once trends have been established, they can then be compared to trends in land use activities for possible correlations.

For example, a broad examination of anadromous fish stocks in the Pacific Northwest and California was conducted by Nehlsen et al. (1991) to identify stocks of a high or moderate risk of extinction or that are of special concern. Recent spawning escapement trends for seven anadromous salmonid species were utilized to assess stock risk and the major factors threatening these at-risk populations identified in published literature and by professional judgment. They identified 214 native, naturally spawning stocks of which one was classified as threatened under the ESA and as endangered by the State of California, 101 were at high risk of extinction, 58 at moderate risk of extinction, and 54 of special concern. A summary of at-risk populations by species followed. One finding was that native fall chinook (*Oncorhynchus tshawytscha*) stocks were under particular pressure on the Oregon coast and in the Snake and lower Columbia River basins. They specified that the native upriver fall chinook population in the upper Columbia was strong within the Hanford Reach, WA (Huntington et al. 1996 also classify this as the farthest inland population in healthy condition), while native, naturally spawning populations had declined to very low levels within the Snake River. Dauble and Watson (1997) further refined the differences between these two systems using fish passage counts at individual dams to examine escapement above McNary Dam. They estimated that the use of the Hanford Reach by fall chinook for spawning increased from 60% of the total run above McNary Dam in the 1960s to 80% of the run in late 80s and early 90s. In contrast, the proportion of the run entering the Snake River declined over this period from 40% to less than 5%. This decline in the mid 1960s and 1970s was attributed to the losses of juveniles passing through turbines and delays of migrations in reservoirs (Raymond 1979).

Once population trends have been identified, it is possible to investigate how these trends might be shaped by both the underlying habitat and habitat change. For example, Dauble and Geist (2000) examined the spawning habitat characteristics of the Hanford and Hells Canyon (Snake River) sites to assess how hydroelectric development has influenced spawning habitat availability. They found the Hanford Reach to be an unconfined, wide, low gradient reach possessing a stable channel bed, porous bed materials, an alluvial flood plain, and generally consistent flows. In contrast, the Hells Canyon Reach is found within an 1800m deep v-shaped gorge with a narrow valley bottom. As a result, this more confined, steep reach possesses bedrock substrate, an unstable channel bed, little small substrate for spawning and intragravel flow, and more variable flows. Spawning sites were linked to the presence of channel bars and channel width in both reaches and lateral bar formation in Hells Canyon. Redd characteristics between the two populations show redds to be found across a greater range of depths and dominant substrate size in the Hanford Reach, while redds from both areas were generally found at velocities between 60 and 120cm/s. They concluded that the Hanford Reach population has remained viable largely due to a geologic template that is highly compatible with their life-history requirements. In contrast, the Hells Canyon population must contend with poor habitat quality and quantity coupled with the elimination of upstream and downstream populations through migration blockage and habitat inundation associated with dam construction.

The focus of abundance examinations is to identify spatial and/or temporal trends in abundance and to correlate these trends with landscape features or land use activity across or within watersheds. However, the resulting correlations cannot identify causation, but rather, highlight detailed investigations needed to uncover the actual processes being impaired and, subsequently, the recovery efforts needed.

Productivity

The productivity of a population can be viewed in terms of sustained trends in abundance, which can be assessed by examining population growth rate, i.e., production realized over the entire life cycle (McElhany et al. 2000). These assessments can be conducted by identifying sustained temporal patterns in abundance or functional relationships that describe the dynamics of a population (e.g., carrying capacity), with the assumption that systematic changes, rather than stochastic events, influence productivity (McElhany et al. 2000). In examining how habitat changes could alter the productivity of a population, it can be useful to concentrate on investigating stage-specific carrying capacity, for example, the smolt production sustainable by the environment, and any habitat characteristics that could be limiting this realized carrying capacity.

It is possible to examine smolt production at a regional level. For example, Bradford et al. (1997) compared coho (*O. kisutch*) smolt abundance data for 86 streams in western North America to habitat variables with the hopes of creating predictive correlations. Smolt data extended back into the 1930s and, due to the scope of the study, historical reconstructions of land use activity was not included in the investigation. However, they did utilize habitat variables that were readily available from maps or discharge records: stream length, valley slope, latitude, and discharge. Results indicated that smolt abundance was strongly correlated with stream length, while smolt production (smolts/km of stream) was correlated with latitude: production lowest and least variable at the edges of the latitudinal range while highest and most variable in the middle latitudes (48 and 50°N). Also, larger age-1 smolts were found in longer streams in broader floodplains. They concluded that smolt production is limited by the availability of physical habitat; however, annual variation in smolt abundance was substantial and confidence limits around a prediction of smolt yield for a single stream were very wide. Similarly, Bradford (1999) used the same smolt dataset to examine the role of freshwater conditions in shaping smolt abundance trends. He found that 1) consistent covariation in coho smolt abundance extends only to populations less than 30km apart (i.e., within the same watershed), 2) long-term trends in smolt abundance were weak relative to interannual variability, and 3) interannual variation in smolt abundance was positively correlated among neighboring populations. Thus, both these large-scale studies suggest that it is smaller-scale influences that shape smolt production and that site-specific, data-intensive habitat models are required for evaluating coho production at this scale.

Nickelson et al. (1992) used this smaller-scale approach to estimate coho smolt production for coastal Oregon basins using juvenile density estimates by habitat type for different seasons. Fully seeded streams were sampled each season and habitat was classified using a modified version of the habitat classification scheme described by Bisson et al. (1982). A total of 15 streams and 150 habitat units were sampled and density estimates were generated for each habitat type. Juvenile coho were most abundant in pool habitat throughout the year, but the type of pool utilized varied with season. During spring, fry were predominantly in backwater pools, while in the winter parr densities were highest in alcoves and dammed pools (predominantly beaver ponds). As alcoves and beaver ponds composed only 9% of the total number of habitat units sampled and 31% of the winter area sampled, they concluded that production of smolts in Oregon coastal streams can be limited by the availability of winter habitat.

Once such bottlenecks are identified, they can guide the recovery planning process toward measuring the quantity of critical habitats available in non-impacted versus impacted sites. Also, historical reconstructions can be conducted to quantify the actual losses of these critical habitats in impacted areas. For example, Beechie et al. (1994) estimated the magnitude of lost rearing habitat, and the subsequent loss in coho smolt production, by habitat type and form of impact within the Skagit River Basin. They utilized a combination of field surveys, topographic and inventory maps, and orthophotos to estimate current and historic habitat. Habitat specific useable area factors, rearing densities, and survival to smoltification rates from Reeves et al. (1989) were used to generate smolt production estimates. Results showed a decrease in smolt production capacity of winter rearing habitats of 34% from historical production (1.17 million smolts from a historic 1.77 million), with the largest loss of habitat in side-channel and distributary sloughs mostly within the floodplain and delta areas. Hydromodification, largely due to diking in order to protect lands zoned for agricultural, rural residential, or urban uses, accounted for 91% of the total smolt production losses for winter rearing areas. This type of information is useful in the recovery process, for it can highlight the processes that need to be restored to improve the health of the local habitat as well as set recovery priorities based on the degree of degradation. See the "Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions" and "Prioritizing Potential Restoration Actions Within Watersheds" sections for a more detailed examination of this process.

Diversity

Diversity refers to the distribution of traits within and among populations (McElhany et al. 2000). Genotypic and phenotypic diversity occur at all scales, however, the majority of genotypic diversity is contained within stocks while most phenotypic diversity is greater across populations and landscapes (Healey and Prince 1995). Thus, successful conservation must focus as much on ensuring habitat quality and connectivity as on genotypes (Healey and Prince 1995). The genetic controls are beyond the scope of this document, but it is possible to examine how habitat changes influence the phenotypic expression of traits. Differences in these traits can have adaptive value and should be maintained, even if they don't have a genetic basis, and can be expressed as changes in morphology, fecundity, run timing, spawn timing, behavior, smolt age, age at maturity, and egg size (McElhany et al. 2000). For example, in Carnation Creek, British Columbia, a decline in 0+ coho densities and an accompanying increased growth rate following logging resulted in greater rates of 1+ smolt production (Hartman et al. 1996). Prior to logging, 50-75% of the smolts were age 1+ and the rest 2+, while after logging 95% were age 1+ (Holtby and Scrivener 1989). Age 2+ smolts have higher marine survival, thus this shift in the age and size composition of smolts resulted in an increase in the variability of adult production (Holtby and Scrivener 1989). Unfortunately, long-term databases that can illustrate the results of anthropogenic activities are rare. Thus, other ways to determine widespread threats to diversity are needed.

One approach is to conduct a regional examination of changes in species diversity under the assumption that where species diversity is decreasing, phenotypic and genetic diversity might also be decreasing. For example, Frissell (1993) expanded on the identification of Pacific salmon stocks at risk synthesized by Nehlsen et al. (1991) to map region-wide patterns in fish diversity. He used data compiled for inland fishes (Williams et al. 1989) and anadromous stocks (Nehlsen et al. 1991), focusing on the Pacific Northwest and California, which highlighted species and stocks at risk. Mapping units were based on a drainage basin size of 50-2000km^2 and the basins categorized according to the number of species (0-1, 2-3, 4-5, or 6-8) classified as extinct, endangered, or threatened. Isopleths were then fitted between the categories. Results indicated a general increase in endangerment from north to south. Of particular concern were regions with 4+ species at risk, including the Clearwater, Spokane, and upper Snake River basins in Idaho, rivers of southern Puget Sound, and the Sacramento, San Joaquin, and Los Angeles

River basins in California. Species declines in these basins were mainly due to large-scale dams and irrigation projects. Also, dams, logging roads, and floods have damaged habitats and severely threatened faunal diversity in the Klamath River basin of northern California and southern Oregon. These types of regional examinations can provide insight into those specific basins that are declining and in need of more detailed examinations to identify the specific processes responsible for the decline and, subsequently, the efforts needed to remedy the condition.

It is also possible to conduct bioassessments to identify where anthropogenic activities have eroded stream health, and subsequently threaten diversity. One method is to assess habitat degradation using invertebrate species assemblages. For example, the reference-condition approach requires the development of a reference database containing invertebrate assemblages and matching habitat descriptors for a large number of minimally disturbed "reference" sites. Invertebrate assemblages at these reference sites are described, classified, and related to habitat attributes to develop predictive models. The resulting reference models can then be compared to test sites to identify impairment. Results will highlight impaired areas, but will not specify the underlying causes. See "Habitat Analyses for Phase II Recovery Planning: Identifying Ecosystem Restoration Actions" on page XX, "Prioritizing Potential Restoration Actions Within Watersheds" on page XX, and "Updating the Recovery Plan" on page XX for a more detailed discussion of these methodologies.

Another method of examining the effects of land use on stream health and diversity is to characterize changes in species assemblages across land use gradients. In affiliation with the National Water-Quality Assessment (NAWQA) Program, Waite and Carpenter (2000) utilized this approach within the Willamette River basin in Oregon to identify how natural and land use gradients influence biological assemblages. Data was collected via a combination of field (fish and water quality sampling, habitat measurements) and GIS (land use, drainage area, elevation) techniques, covering reaches within seven major subbasins and three ecoregions. Results found physical habitat a better descriptor of fish assemblages across ecoregions (i.e., all sites), whereas water chemistry was better within the Willamette Valley ecoregion. All sites could be separated into four stream types based on fish assemblages: 1) coldwater forested, 2) agricultural- or urban-dominated, 3) medium-sized river, and 4) heavily impacted. In the medium-sized and heavily impacted areas high abundances of introduced species were associated with high percent external abnormalities, largely due to high water temperature, low DO concentration, and low physical habitat diversity (and somewhat due to increased concentrations of nutrients and pesticides). They suggested that the reduced riparian quality and increased water temperatures, nutrient, and sediment supply found in small agricultural and urban streams cause fish assemblages to shift from those dominated by native, intolerant species to those dominated by introduced or tolerant species. Amount of riffle habitat, the quality of the riparian cover, and the maximum water temperature were the overriding variables describing variances among land uses. These biological assessments provide an opportunity to monitor long-term changes in community composition and/or stream health (Rabeni 1992) at multiple scales, highlighting areas of possible impairment due to land use activities, and identify possible causes that can then be further investigated.

Spatial Structure

The spatial structure of a population refers to the spatial distribution of individuals within a population and the processes generating that distribution, and is dependent on habitat quality, spatial configuration, and the dynamics and dispersal characteristics of individuals in the population (McElhany et al. 2000). As the stream environment is heterogeneous, it may be viewed as a series of “habitat patches” at an array of spatial scales. As habitat quality varies across patches, the likelihood of individuals inhabiting each patch is dependent on the quality of the habitat in the patch as well as the ability of individuals to move between patches. Therefore, a proper investigation into how habitat changes might have altered the spatial structure of a population requires understanding the small- and large-scale influences on both habitat patch dynamics and salmonid movement.

One starting place is to examine historical presence of a species and compare it to predicted or current occurrence. This can identify the distribution of a species across the landscape identifying populations, their degree of isolation, and their size. Rieman et al. (1997) looked at potential historic and known distribution of bull trout (*Salvelinus confluentus*) across 4,462 subwatersheds of the interior Columbia River basin, representing 20% of the species’ global range. Over 150 state, federal, tribal, and private biologists across the region summarized available information on the presence and status of bull trout, which was then validated and augmented with state databases. Classification trees were then developed to predict bull trout presence and status using landscape features and management history, and to identify those elements most closely associated with bull trout distribution. Bull trout were widely distributed within their potential range, occurring in about 36% of the subwatersheds, however, much of this distribution was patchy and distributions disjunct. Bull trout were estimated to occur in 44%, and be strong in 12%, of the historic spawning and rearing subwatersheds. Populations were more likely to occur in steeper, higher elevation, mid-order subwatersheds with strongest populations occurring in subwatersheds with mean annual air temperatures less than 5.1°C and containing less than 2.5 miles of road per square mile. So, these coarse examinations can highlight the distribution of populations across the landscape and give some indication of strength based on size. While some of the patterns in distribution can be identified through these analyses, more detailed work is needed to address patch size and dynamics within individual systems.

Dunham and Rieman (1999) examined 720 sites (10m), 179 reaches (102m), and 81 patches ($\geq 10^3$ m) within the Boise River to identify patterns in juvenile bull trout occurrence. Habitat patches were defined as stream catchments above 1,600m elevation with an accessible perennial stream (Rieman and McIntyre 1995), and patches, road densities, and interpatch distances were estimated using GIS methodologies. Trout occurrence, stream width, and gradient data were obtained in the field. Results indicated that the large-scale geometry of catchments can strongly influence the distribution of aquatic species, although smaller-scale factors also affect distribution. Bull trout were present in 36% of potentially occupied habitat patches, and were generally found in larger, less-isolated patches with low road densities. Within these occupied patches, bull trout were found in streams wider than 2 meters. They concluded that bull trout populations in larger, less isolated, less disturbed patches are more likely to persist and that it is critical for disturbance within these habitats to be minimal. On the other hand, small, isolated, disturbed populations and habitats are at risk. Thus, they speculate that conservation and restoration opportunities might meet best results if centered within those patches of intermediate size and/or isolation. In a comparison of patch size distributions for Lahontan cutthroat trout (*O. clarki henshawi*) and this bull trout data, Dunham et al. (2002) found that the size distributions were similar between the two species and skewed toward smaller patches, so that very few large patches may be critical to each species. Both species are likely to occur when patch size exceeds 10^4 ha in area, and bull trout are more likely to occur in smaller patches that may, in turn, explain their occurrence in a large percentage of suitable habitat.

Identifying how species are distributed across the landscape and utilize habitat patches is a necessary first step in the recovery planning process. Linking this information to their dispersal and migratory behavior can highlight “stepping stone” patches: critical patches connecting large habitat areas. Once identification of critical habitat patches is achieved, it is possible to examine how land use actions have reduced or eliminated connectivity of these patches as well as altered or destroyed the patches themselves. This information can then dictate the scale of recovery efforts by pinpointing the processes that are in need of restoration and by identifying local, specific recovery actions.

Putting It All Together

The foregoing was an attempt to illustrate how the effects of habitat change on the four components of population viability can be examined over multiple scales. However, the components are not exclusive and information gleaned for one component is often useful for revealing information about another. This section concludes with an attempt to illustrate, in a general sense, how to bring all the components together, incorporating scale.

The first step in the process is to identify patterns. This will generally involve the examination of abundance and/or distribution, incorporating any changes over time and/or space. Results of these examinations can often be applied to several of the recovery goals. For example, the work done by Rieman et al. (1997) in the Columbia and Klamath basins and by Dunham and Rieman (1999) in the Boise River basically looked at distributions of bull trout across the landscape, and in the former, predicted historic distributions. In identifying these distributions, each study was able to address the increased patchiness of bull trout populations over time, addressing spatial structure, and to identify larger and smaller populations. In turn, each was able, in association with spatial isolation, to estimate the relative strength and diversity of individual populations and the corresponding degree of risk. Similarly, Thurow et al. (1997) examined the distribution of other native salmonids within the interior Columbia River and portions of the Klamath and Great Basins to determine their distribution and status. Their work highlighted the proportion of the potential range currently occupied by each species and the relative strength of the populations. They found all taxa to have narrower distributions, fewer areas with high diversity, and lower percentages of strongholds than in their estimated potential historical conditions. Strongholds were generally found to be rare and not well distributed across the landscape, with the largest remnants of high salmonid diversity in the central Idaho mountains, the Blue Mountains, the northern Cascades, and their connecting river corridors. Thus, examining abundance and distribution across space and time can illustrate areas retaining historic diversity and ecological structure as well as those that have possibly been altered by human influence.

Scale is important within this framework for it sets constraints on results obtained and on any subsequent interpretations (Wiens 1986), and with increasingly smaller scales more detailed information can be attained. For example, the larger examinations just illustrated highlight systems that warrant further examination, and broadly correlate factors (land use, climate, other species) to population trends. However, it is a more specific examination of particular components of a system (i.e., at watershed or smaller scales) that can identify causation (e.g., road density, hydromodification, water quality degradation). Once identified, and with an understanding of the literature and the area of interest, these causes allow for the detection of specific ecological process impaired by these activities and point toward the best possible course for restoring these natural processes.

An example can be seen for steelhead (*O. mykiss*) within the Eel River in California. In their assessment of anadromous stocks at risk in the Pacific Northwest and California, Nehlson et al. (1991) noted a decline in summer steelhead within most river systems in California. In particular, they identified the Eel River population to be at moderate risk of extinction and mentioned how floods occurring in 1964 severely affected populations throughout California due to extensive erosion and habitat damage in watersheds stressed by poor land management. In an investigation of native and nonindigenous fish within tributaries of the Eel River, Harvey et al. (2002) highlighted the specific mechanisms contributing to these declines. A potential predator on, and competitor to, native salmonids, Sacramento pikeminnow (*Ptychocheilus grandis*) was introduced into the river around 1980, eventually becoming the most abundant species in many parts of the drainage. Young-of-the-year steelhead and pikeminnow are the dominant age-class for each species (79% and 85%, respectively) and abundances of each were predicted well by temperature (maximum weekly average temperature). Age-0 pikeminnow are found in the warmest tributaries and age-0 steelhead in the coolest; however, this habitat separation is not due to physiological tolerances, and temperature-dependent interactions are presumed. They speculate that anthropogenic activities within the drainage allowed for more widespread habitat damage following floods during the 1950s and in 1964. These floods resulted in widespread landslides, a loss of riparian vegetation, and altered summer temperature regimes, which then allowed for the extensive invasion of pikeminnow decades later. The investigators concluded that (1) restoration of riparian vegetation within the watershed could reduce the range and ecological impact of the pikeminnow, and (2) increased riparian vegetation and improved hillslope conditions could also enhance native salmonid habitat by increasing channel stability and influencing peak flows and sediment supply, thereby effecting thermal regimes and instream habitat (latter important for post-age-0 steelhead). Thus, results of larger-scale work identify a problem, while small-scale work highlights the specific anthropogenic activities and ecological impairment underlying the problem and, consequently, more specific conservation and restoration strategies.

As aquatic ecosystems are arranged in an interconnected array of hierarchical systems, any study of their processes and inhabitants can be organized in a similar hierarchical manner. Species patterns and relations to habitat and anthropogenic activities can be seen at any scale; however, the scale examined dictates the level of detail inferred for any results obtained and any subsequent interpretations. It is by examining the abundance and distribution of species and associating these with land use activity at multiple scales that will highlight the spatial structuring and diversity of populations, and thus their level of risk. Such information can then be used to identify the ecological processes impaired and, hopefully, to prioritize conservation and restoration activities.

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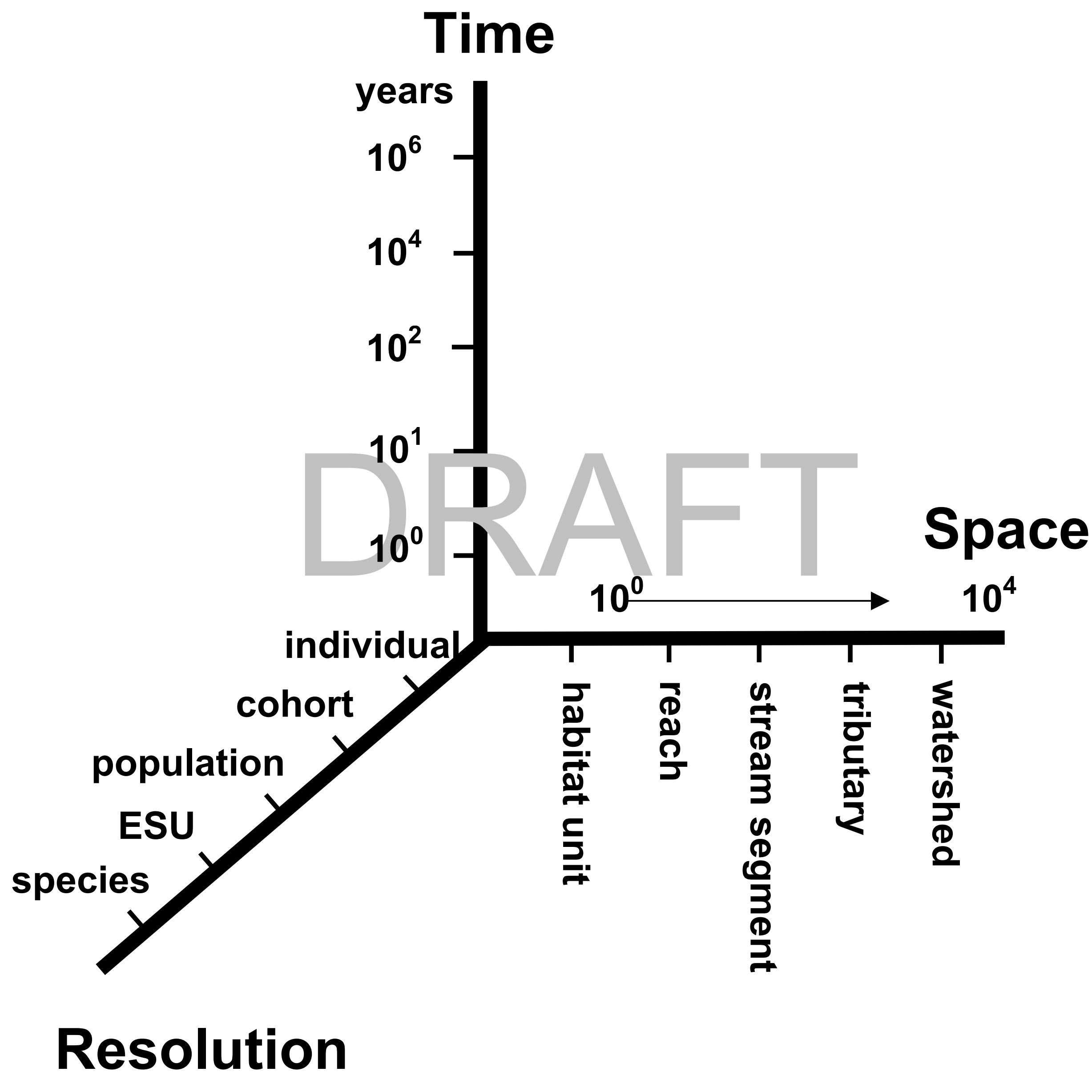
Figure 6-1. Three aspects of scale in aquatic systems (adapted from Frost et al. 1988 and Frissell et al. 1986). In this figure, space identifies the linear spatial scale of the different components of stream systems while time indicates the potential persistence of the different components. The degree of resolution identifies ecological organization (e.g., taxonomic or functional groupings) important in salmonid life-history and restoration. For each axis, scale increases outward from the middle of the plot.



*****FOR REVIEW ONLY*****

MANAGING UNCERTAINTY IN HABITAT RECOVERY PLANNING

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To know one's ignorance is the best part of knowledge

Lao Tzu, The Tao, no.71

Salmon recovery planning requires a complex series of decisions about habitat actions despite large amounts of uncertainty in the available information. This uncertainty can result in risks to habitats and populations from inappropriate management advice (Fogarty et al. 1996). Past failures of management plans to prevent population declines and collapse are in part due to the failure to recognize uncertainty in available information and to a lack of procedures for including uncertainty in the decisionmaking process (Wade 2001). Inevitably, decisions will be based on a tapestry of models, estimates, expert opinion, myth, political posturing, predictions, and data. By identifying, quantifying, and acknowledging the uncertainty in information used for recovery planning, we can increase the likelihood that recovery plans will be successful. The benefits of explicitly accounting for uncertainty include capturing all the available information regarding uncertain factors, providing the full range of possible outcomes and the probability of observing each, and identifying the key drivers of overall uncertainty in model projections (Mishra 2001). This section provides guidance, via quantitative and qualitative examples, for managing uncertainties inherent in habitat recovery planning.

A quick example illustrates how identifying and quantifying uncertainty can help a manager make explicit trade-offs between potential positive outcomes and acceptable risks. In choosing between two possible culverts for restoring fish passage, one might be presented with information that removal of culvert A is predicted to increase fish capacity by 120 fish while removal of culvert B is predicted to increase fish capacity by 100 fish. With no estimates of uncertainty, the manager would surely choose culvert A because it has the highest expected increase in fish capacity. Such a decision would be based on incomplete information. One also needs uncertainty estimates, indications of how likely it is that the true outcome will be close to the point estimates of 120 and 100 fish. More complete information might indicate that replacement of culvert A would open habitat that was less certain to be occupied (120 +/- 70), while replacement of culvert B would open wetland habitat that would be quickly colonized with a high degree of certainty (100 +/- 10). With the more complete information, decision makers could then explicitly choose between a higher but less likely increase in fish capacity and a lower but more certain increase in fish capacity. In this example, both actions were unlikely to cause harm (a negative change in fish capacity). In other situations, actions with a high potential pay-off may also contain some risk of being detrimental to fish, for example, when deciding whether to use chemical herbicides to remove non-native vegetation from riparian areas. Without an estimate of the magnitude of the uncertainty in information on which decisions must be made, decision makers cannot make informed decisions.

The importance of clearly communicating uncertainty has been repeatedly emphasized in the fisheries literature (Francis and Shotton 1997):

- “Understanding the risk or uncertainty associated with choices could help fisheries managers select management strategies, decide which types of risks and uncertainty inhibit the effectiveness of management techniques, and finally, recognize which types of uncertainty must inevitably remain…" (Peterson and Smith 1982);

- "Point estimates should be accompanied by variance estimates." (USCTC United States Chinook Technical Committee 1997);
- "The managers' task may be made easier if uncertainty in a fishery assessment were expressed…" (Francis 1992);
- "Scientific advice to fishery managers needs to be expressed in probabilistic terms to convey the uncertainty about the consequences of alternative harvesting policies." (McAllister et al. 1994);
- "Clearly, when management decisions are to be based on quantitative estimates from fishery assessment models, it is desirable that the uncertainty be quantified, and used to calculate the probability of achieving the desired target and/or risk of incurring undesirable events." (Caddy and Mahon 1995).

Reporting uncertainty in data and predictions has become common in harvest management (Rosenberg and Restrepo 1994). Uncertainty is not as often incorporated into decisions about salmon recovery planning and habitat management despite broad consensus that it is important and necessary to consider uncertainty in the conservation and management of species (Mangel et al. 1996, Flaaten et al. 1998, Akcakaya et al. 2000, Ralls and Taylor 2000, Wade 2001).

Additional information and improvements in decisionmaking can be gained by partitioning the uncertainty into components such as measurement error versus model uncertainty. In the culvert example, partitioning of the uncertainty might reveal that the most significant source of uncertainty about culvert A is insufficient data on juvenile use of a few habitat types. By identifying knowledge gaps that limit decisionmaking we can prioritize information gathering and model development. We emphasize that all information used in habitat analyses should include, at a minimum, both the observed or predicted value and an estimate of the associated uncertainty.

In this section, we first describe five types of uncertainty embedded in a prediction of habitat capacity. We follow this with two examples of uncertainty in habitat management issues related to recovery planning. In each example, we describe how management decisions might be improved by acknowledging, quantifying, and reducing uncertainty in the decisionmaking process. The first example describes qualitative strategies for reducing uncertainties regarding chemical contaminants and for making structured decisions in the face of limited empirical data. The second example describes the use of decision tables for making decisions that incorporate uncertainty. The final subsection describes strategies for making decisions when empirical data are lacking. In this subsection, we make a distinction between variability, characterized by true differences in the value assumed by a variable over time, space, or populations, versus uncertainty, or a lack of knowledge about a true and constant value (Morgan et al. 1990, Cullen and Frey 1999). Our discussion of methods for reducing uncertainty is purposefully simplified throughout, but references are provided for each example so that interested readers can locate more detailed information. We hope that by deleting site-specific and mathematical details, a general framework for incorporating uncertainty into decisions can be expressed.

Types of Uncertainty

Precise and accurate predictions are a fundamental goal in the aquatic sciences. Improved management of aquatic resources will result from a predictive science that can forecast the consequences, costs, and benefits of management actions (Pace 2001). A prediction might be a predicted value (e.g. habitat capacity estimate, extinction risk, or survival rate) or a predicted relationship between a habitat action and a biological response (e.g. effects of high flows on egg survival, effects of a particular restoration technique on fish survival, or projected population trajectories under different climate scenarios). Population viability and habitat goals as well as prioritized project lists and watershed plans must be developed from these types of predicted values and relationships. Informed plans and decisions will be based on both the predictions and the uncertainty surrounding them. In this section, we describe five types of uncertainty found in predictions of habitat capacity: predictive uncertainty, parameter uncertainty, model uncertainty, measurement uncertainty, and natural stochastic variation (Table 7-1).

Estimates of uncertainty are critical to informed decisionmaking. Evaluating the relative magnitudes of the five classes of uncertainty embedded in a particular prediction is valuable because it tells us where to be skeptical. More formally, we may pursue value of information (VOI) analysis to establish which additional information is most likely to improve our decisionmaking position (Raiffa and Schlaifer 1961, Raiffa 1997). VOI techniques seek to identify situations in which the cost of reducing uncertainty is outweighed by the benefit of the reduction. In some cases, it may turn out that the predictive uncertainty is prohibitively large and, therefore, that the available empirical data provides little guidance for decisionmaking. In such cases, other decisionmaking processes that do not require quantitative predictions can be used.

To a great degree, the five components of uncertainty are nested: prediction uncertainty includes parameter and model uncertainty, which each includes measurement error and natural variability. We provide examples of how each type of uncertainty arises, how it might be quantified, and how it might be reduced (Table 7-1). We conclude each subsection with a summary of how decisionmaking can be improved by quantifying and acknowledging each class of uncertainty. A series of questions to ask of any prediction is provided in Table 7-2.

Prediction Uncertainty

Predictions include uncertainty from natural stochastic variation of the system being modeled, measurement uncertainty of the data used to build the model, uncertainty surrounding the form of the model, and parameter uncertainty (components addressed in the following subsections). In addition, predictions can also include uncertainty that results from applying a model to a new situation. A capacity estimate for Watershed X might predict future capacity based on current and past data for the same watershed. Or, current predictions for Watershed X might be based on data collected in other watersheds. Both cases involve extrapolating from conditions under which data were collected to new conditions of interest. Uncertainty associated with these or similar extrapolations from, say, the laboratory to the field, is difficult or impossible to quantify but must be considered and described.

For predictions that do not require extrapolation, prediction uncertainty can be evaluated by “ground-truthing”, by prediction confidence intervals, and by cross-validation simulation studies. Ground-truthing will help quantify the accuracy and precision of past predictions about current conditions, but can only suggest how well the model may perform under future conditions. Prediction confidence intervals can be computed in situations for which the manager does not need to extrapolate beyond the original data (Zar 1984). Where there is more than one predictor variable, caution should used in defining the joint sample space beyond which one is extrapolating. In cross-validation simulations, the model is constructed and parameterized using a subset of the data (Stone 1974). The model is then assessed by how well it predicts that subset of data excluded from model construction. Cross-validation simulations do not include uncertainty associated with extrapolating from measured to unmeasured conditions. To assess how well a model may predict unmeasured conditions requires careful consideration of those model components that may be sensitive to expected differences between measured and unmeasured conditions (i.e., current versus future conditions). Models can be compared in their relative sensitivity to changing conditions. Models that rely on predictors that are only correlated with the causal factors are particularly likely to have high levels of prediction uncertainty because in new situations the correlations on which the model is based may no longer be coincident with the causal mechanism.

Parameter Uncertainty

Model parameters are necessarily estimated with uncertainty. A statement of the uncertainty of these parameter estimates is critical for making informed management decisions. For example, if instream wood restoration were estimated to increase fry to smolt survival within the treatment reach by 20%, we might embark on a widespread wood placement plan. However, if the estimate were more completely expressed as 20% +/- 30%, we might diversify the types of restoration actions used, or choose a different restoration action with a smaller but more certain fish response and little or no risk of an adverse affect.

Parameters that have biological meaning provide a context for interpreting the associated uncertainty. For statistical models, parameter estimates are developed from the data and the uncertainty associated with these estimates is relatively easy to compute. For mechanistic models, parameters may be estimated from data, from similar models of other phenomena, or by expert opinion. Where parameters are not estimated from data, the uncertainty surrounding them can be difficult or impossible to quantify.

Sensitivity analyses can be used to estimate the effect of parameter uncertainty. Nominal range or local sensitivity analysis computes the effect on model outputs of systematically varying each of the parameters in the model across its range of plausible values while holding the other inputs at their nominal values. Where small changes in parameter values lead to large changes in model predictions, it is important to reduce the uncertainty of those parameter estimates whenever the value of additional information outweighs the cost of obtaining it. Models that are extremely sensitive to changes in parameter estimates and which have highly uncertain estimates of those parameters will yield predictions with large uncertainty. Even where models produce highly uncertain predictions, they may be useful in quantifying the uncertainty in predictions and in determining the type and quality of information that would be required to produce predictions with acceptable levels of certainty. The sensitivity analysis tells the manager that predictions are very sensitive to particular conditions and that they will either have to increase precision of parameter estimates or ensure that management plans are robust to expected uncertainty. Increased precision of parameter estimates can be achieved by collecting more data, collecting data over a wider range of values, or collecting better data (data with less measurement uncertainty).

Model Uncertainty

Nearly all estimates and predictions used in management are based on an underlying model, either explicitly or implicitly. Uncertainty exists about both the model form (for example a linear relationship versus a Ricker curve) and about which predictor variables to include.

Model uncertainty results from an incomplete understanding and a simplified representation of ecological systems and functions (Fogarty et al. 1996). In our capacity example, we might have a model that predicts habitat capacity as a linear function of several habitat parameters: wood density, pool density, gradient, adjacent land use, and water temperature. The default assumption may be to use a simple linear regression model. However, there is uncertainty as to whether the effects of these five habitat descriptors are additive or have a linear relationship to habitat capacity. The linearity assumption may be valid for the range of, say, water temperature for which we have data, but invalid outside that range. We are also unsure if these five habitat descriptors are the best set of predictors or if an alternate set of predictors might perform just as well. Many statistical tools are available for choosing between models (adjusted R-squared, AIC, BIC, F-tests, likelihood ratio tests, cross-validation metrics) (Burnham and Anderson 1998). In general these techniques balance the degree to which the model fits or predicts the data with the complexity of the model (usually expressed as the number of parameters).

Models that fail to accurately describe the ecological process or include an important predictor can have enormous management implications. Model predictions can be of the wrong magnitude or even the wrong direction. Managers and ecologists have often erred significantly by failing to consider model uncertainty. For example, the prevailing model of habitat effects on fish survival once assumed that fish survival decreases with increasing amounts of in stream wood, and as a result large amounts of wood were removed from streams and rivers (Maser et al. 1988). Thus, habitat degradation in the Pacific Northwest can in part be attributed to a failure to assess the possibility that this model was incorrect (Beechie et al. 1996).

Model uncertainty is very difficult to quantify because there are an infinite number of possible models; none is exactly correct. Simulation studies generate data using a particular model and then ask questions about the behavior of those data (Morgan et al. 1990). They can quantify the degree to which the structure of the model influences the model's predictions. Averaging predictions from a suite of models can reduce the impact of model uncertainty on management predictions (Burnham and Anderson 1998, Cullen and Frey 1999). Beyond these tools, reducing model uncertainty is extremely difficult. Schnute and Richards (2001) suggest that model uncertainty be managed by keeping an open mind, identifying all assumptions, and testing assumptions continuously.

Measurement Uncertainty

Measurement uncertainty, or observation error, is simply the difference between a true value and our recorded observation of it. It results from measurement, sampling, and data processing errors (Francis and Shotton 1997). All observations carry some degree of measurement uncertainty. This uncertainty may be large and problematic or small and of negligible consequence. Some phenomena, such as the survival of fish in different habitats, are inherently difficult to measure. Consequently, the variables associated with these phenomena have a high degree of measurement uncertainty. Other phenomena, such as stream discharge, can be measured quite accurately. Uncertainty resulting from sampling error occurs when the measured units are not representative of the population for which inference is being made. The incorporation of measurement and sampling errors can obscure or create relationships between variables (Ludwig and Walters 1981, Walters and Ludwig 1981). Measurement error, as defined here, can also occur during data processing and storage.

Measurement uncertainty is directly related to both the accuracy and precision of the measurement technique. Accuracy, the inverse of uncertainty, in a measurement technique describes the average distance between the measured value and the truth. The precision of a measurement describes the variability around that average. Therefore, it is quite possible for a measurement tool to be highly precise (very low variance across repeated measurements) and yet inaccurate (the average of repeated measurements is far from the true value). In other words it is possible for a measurement to be characterized by very little variability but by a large degree of uncertainty. While there have been many attempts to estimate measurement uncertainty in, for example, habitat surveys (Pleus 1995, Roper and Scarnecchia 1995, Poole et al. 1997) or redd surveys (Jones III et al. 1998, Dunham et al. 2001), the known uncertainty in these types of data is rarely included in the uncertainty of predictions from models that are based on these types of data.

Measurement uncertainty can result in systematic error or bias. Bias is a directional error that results from measurement using a systematically inaccurate tool. Biased or potentially biased measurements might include subjective assessments or incomplete records. A less visible form of bias occurs when a measurement technique tends to overestimate in certain conditions and underestimate in other conditions. A simple example is helicopter redd surveys. Redds are easier to identify where there are fewer trees; therefore the accuracy, or uncertainty, of the measurement depends on whether there are riparian buffers. If the bias is not corrected, the data can erroneously predict increases in redd density with removal of riparian trees.

Measurement uncertainty can be reduced but not eliminated. Replication is the best way to reduce the uncertainty though it will not remove bias resulting from the use of inaccurate measurement tools. The best way to manage bias is to estimate it using at least some unbiased measurements and then correct for it. In some cases, measurement uncertainty can be very difficult to assess. Expert opinion or subjective assessments, for example, are often used because no actual data exists. Although it may be possible to determine how well experts agree with one another (precision), it is impossible to assess or quantify the relevant issue (accuracy) when there are no accurately measured data available. Measurement uncertainty may also be quantified using repeated measurements or by computer-intensive techniques such as resampling or bootstrap methods (Efron and Tibshirani 1991, Efron and Tibshirani 1993).

By quantifying measurement uncertainty, the value of collecting more data with the same measurement or sampling technique versus a more expensive technique can be weighed. Where bias is impossible to measure or quantify, sensitivity analyses (as described in the subsection on parameter estimation uncertainty) can provide an assessment of the degree to which small amounts of measurement uncertainty or bias in the input data might effect predictions (Morgan et al. 1990).

Natural Stochastic Variation

Natural stochastic variation is the inherent random variability in ecological systems, such as temperature or population fluctuations. It also incorporates the underlying stochastic nature of population dynamics (Rosenberg and Restrepo 1994). It contributes to our inability to make precise predictions. Increased amounts of natural stochastic variation, often called process uncertainty, require increased numbers of observations (either more sites or more replications or both) to make estimates of a given precision (Shea and Mangel 2001). Very high levels of natural variation can mean that estimates of the required precision are simply impossible to obtain (Korman and Higgins 1997).

Identifying and quantifying natural stochastic variation helps us to distinguish between situations in which small amounts of additional data should dramatically increase our ability to make good decisions, and situations in which additional data are unlikely to provide significant increases in the accuracy of predictions. This is the heart of value of information analysis as discussed earlier. While we may be able to improve on our knowledge of the true value of an 'uncertain' parameter, we can only improve on our understanding of the behavior of a 'variable' parameter. In some cases, stratifying the data or redefining the question can reduce the effects of natural stochastic variation. For example, we might make separate estimates of in-river survival for wet versus dry years. Managers would then be able to make more informed decisions about the value of habitat restoration plans that potentially have different effects in wet versus dry years. Because stochastic variation is a natural phenomenon, it cannot be reduced to increase the precision of our predictions. Where it can no longer be reduced by stratification, stochastic variation is best managed by simply quantifying and acknowledging it.

In summary, an informed management decision requires information about the uncertainty of the predictions on which that decision will be based (Pace 2001, Regan et al. 2002). Evaluating the uncertainty in each prediction requires the dissection of that uncertainty into its components—quantifying and evaluating each and then recombining them into a single measure of overall uncertainty. Each class of uncertainty as well as methods for quantifying and reducing it are summarized in Table 7-1. By asking the questions in Table 7-2, we reduce the chances of making poor or uninformed decisions because of poor predictions. Reagan (2002) has identified further types of uncertainty that she describes as linguistic uncertainty, including concepts such as numerical vagueness, context dependence, ambiguity, and underspecificity. While we do not consider these forms of uncertainty explicitly in this section, they are certainly common in habitat science and should be sought out, evaluated and treated (Regan et al. 2002).

Uncertainty analysis is key to identifying critical knowledge gaps, identifying options for improving predictions, and making the most informed habitat decisions.

Example 1: Water Quality and Salmon Health, Uncertainty Across Multiple Scales

Uncertainty in habitat planning can result from many sources including the omission of a key habitat variable or mismatches between the scale at which data is collected and the scale at which the information is applied. In this subsection, we provide qualitative suggestions for managing these problems, using water quality as an example. The quantity and quality of salmon habitat are both important determinants of salmon productivity and population viability. Stream temperatures, sedimentation, and water pollution are all examples of habitat quality. Importantly, however, empirical data for the various forms of water pollution are rarely incorporated into habitat models. Consequently, the complex impacts of urbanization, agricultural land uses, and industrial activities on the chemical condition of salmon habitat may not be considered. As we move to decisions about which habitat actions should receive the highest priority and where actions should occur, issues of habitat quality become critical in making good choices. This is particularly true for habitats that have mixed physical and chemical degradation. In this example, we describe water quality issues that are directly relevant to salmon ecology, explain discontinuities between the scales of ecotoxicological studies and the informational needs of recovery planners, and suggest ways to improve habitat decisionmaking by incorporating water quality data. In exploring this example, we provide non-quantitative solutions to reducing some types of uncertainty that are common in habitat recovery planning.

Water Quality and Salmon Habitat

Pollution is an important habitat concern for the vast majority of threatened and endangered salmonids in the Pacific Northwest. Environmental monitoring studies have consistently detected a wide array of metals, pesticides, and other toxic substances in the surface water and sediment of salmon habitats, and also in the tissues of salmon themselves. Despite documented exposure conditions (Wentz et al. 1988, Ebbert and Embrey 2002), the impact of environmental contaminants on salmon health or on the biological integrity of aquatic systems is poorly understood. Even given that uncertainty, habitat-based models for salmon recovery must capture the biological significance of water and sediment quality.

Water quality may affect salmon abundance and survival via direct or indirect effects. Pollutants may have immediate lethal effects on individual fish. However, such effects are rare compared to the vast array of potential sublethal effects that may reduce individual fitness and population performance. Chemical habitat quality can also impact individuals and populations indirectly via reductions in the abundance of key prey taxa. Predictions of salmon productivity are likely to have high levels of model and prediction uncertainty if the specifics of water and sediment quality are not included in model development.

There are several reasons why the specific determinants of chemical habitat quality are often excluded from habitat models. First, “chemical habitat quality” can be difficult and expensive to measure. Accidental releases of large quantities of a single lethal contaminant are rare and unpredictable. More often, recovery planners must contend with diffuse (or non-point source), sporadic deliveries of mixtures of contaminants to salmon habitat, such as might result from urban or agricultural land-use. Exposures to these chemical mixtures are almost always sublethal. That is, fish rarely die from the acute failure of a critical physiological system. However, sublethal effects can lead to so-called "ecological death" (Kruzynski and Birtwell 1994) if they reduce the likelihood that an animal will survive and reproduce. Second, there is a general absence of toxicological data for most of the chemicals that have been detected in salmon habitat. Only a few studies have specifically addressed the impacts of environmental contaminants on biological processes in Pacific salmon that are clearly linked to survival, migratory success, or reproductive success (Kruzynski and Birtwell 1994, Arkoosh et al. 1998, Hansen et al. 1999, Heintz et al. 2000, Scholz et al. 2000, Rice et al. 2001, Meador et al. 2002). Third, effects at the scale of natural populations are rarely considered in conventional toxicological studies. Many conventional endpoints or biomarkers of chemical exposure have no clear or consistent relationship to the survival or reproductive success of the exposed animal. Consequently, there is often a disconnect between the biological scale at which toxicological studies are conducted and the data requirements for current habitat recovery models (Hansen and Johnson 1999b, Hansen and Johnson 1999a).

Incorporating Toxicological Data

It is important that recovery plans capture important broad spatial and temporal patterns of chemical habitat degradation in order to help identify uncertainties around predicted outcomes of restoration actions. Contaminants occur in complex mixtures whose composition varies in time and space, and salmon habitat conditions may reflect current land use activities or activities that were restricted or banned many years ago (e.g., for persistent chemicals such as DDT). Moreover, water quality at a specific point within a watershed may be determined by land use activities that are far removed from the focus of restoration efforts. Acknowledging the large spatial and temporal scales at which contaminants can affect fish helps identify some of the uncertainty associate with predicting the effects of restoration actions. For example, we may have data on chemical contamination, but we do not know how to incorporate it into habitat capacity or survival estimates. Nevertheless, this information might inform decisionmaking as we can surmise that the uncertainty of predicted increases in habitat capacity for a given restoration action are likely higher in areas with high levels of past or present, on-site or upstream chemical contamination. Likewise, we might expect inaccuracy and uncertainty in survival estimates that are extrapolated from a stock within a pristine watershed to a stock that migrates through a highly contaminated estuary.

Uncertainty in habitat recovery plans can be reduced by explicitly incorporating information about chemical water and sediment quality. Recovery planners must address two important questions with respect to chemical habitat quality. First, do water and sediment pollution limit the availability of food for salmon during freshwater and estuarine life history stages? And second, are contaminants having sublethal effects on essential physiological processes or behavioral patterns of exposed fish?

Recovery planners can, in many cases, make informed decisions about the prioritization of water quality improvements versus physical habitat restoration. For example, if the goal of a recovery effort is to restore properly functioning conditions and, by extension, a productive aquatic food chain, the consequences of periodic pulses of insecticides (or even herbicides) can be estimated either quantitatively or qualitatively. In watersheds where insecticides occur (primarily in agricultural and urban areas), it should be possible to estimate the potential loss of invertebrate prey, the subsequent reduction in the growth of juvenile fish, and the likelihood that salmon from contaminated habitats will have a lower rate of marine survival. If environmental monitoring data are unavailable, recovery planners might extrapolate potential chemical concentrations from other (monitored) basins with similar agricultural or urban land use. Even simple comparisons between reported environmental concentrations and toxicity thresholds for aquatic invertebrates can reduce the scientific uncertainty surrounding the potential effects of contaminants on salmon productivity. This, in turn, would improve restoration prioritization and watershed management plans.

The challenge in estimating the direct effects of toxic chemicals on salmon health is to identify (1) which contaminants are known (or suspected) to occur in salmon habitat, and (2) pathways of toxicity for these chemicals that have clear significance for the survival, migratory success, or reproductive success of wild salmon. The potential for sublethal effects on individual salmon that contribute to population declines must be incorporated into habitat recovery plans despite the high levels of associated uncertainty. This mechanistic approach will require that recovery plans consider the relevant toxicological data for the contaminant of concern (e.g., sublethal impacts on the immune, endocrine, reproductive, or nervous systems of salmon).

Habitat models carry a high risk of large predictive errors if they treat pollution as a single variable or a collection of variables that can somehow be compared to individual physical habitat metrics. That is, there is no biological basis for grouping dioxins, metals, and PCBs into one variable because each class of chemicals has different mechanisms of action and effects salmon fitness and survival in a different way. Errors that result from extrapolating results between classes of chemicals with completely different mechanisms of action can be avoided by developing separate mechanistic models for each class of chemical.

Applying new or existing empirical data to update existing mechanistic models can also help reduce uncertainty. Planners or researchers should utilize the primary toxicological literature in the development of recovery plans in order to avoid mandating inappropriate actions in habitats that are unquestionably impacted by various pollutants. Answers to the following questions can often be found in the toxicological literature and will enable more accurate and precise predictions about the effects of specific chemical contaminants on salmon population performance. What is the evidence that a contaminant or class of contaminants is present in salmon habitat? What are the expected environmental concentrations? How long will exposures last? What life history stages of salmon are likely to be affected? What are the primary or known pathways for sublethal toxicity for the contaminant in fish? Do these effects have clear consequences for the health or performance of the fish? And, what are the chances that the contaminant is significantly limiting salmon productivity within the geographical area of concern?

The most informed management decisions will be those that consider all of the available scientific information. For water quality and other habitat characteristics about which less is known, it is clearly better to acknowledge the uncertainties and incorporate the available information, no matter how limited. In the example of water quality, we can estimate and incorporate the direction of the effect even when we are not yet able to quantify the magnitude of that effect. Moreover, identifying key uncertainties will help establish priorities for ongoing and future research.

Example 2: Creating a Prioritized List of Restoration Projects

Once we have a series of predictions with their associated uncertainties, we must combine them into an action plan. In this example, we demonstrate one method for using predictions and their confidence intervals to develop a project list for a habitat recovery plan.

Developing a project list is difficult because of uncertainty about how fish may respond to changes in the environment. For example, we may have a list of potential actions, each of which is expected to increase pool habitat. There are uncertainties in estimating the increase in pool area, and there is uncertainty about the density of fish that can be supported by a given amount of pool habitat. By explicitly including the uncertainty in a decision table, we can identify the actions with the highest expected final fish density and determine the potential value of reducing the uncertainty. Analogous examples have been worked out in the harvest literature (Hilborn and Walters 1992).

The first task in setting up a decision table is to describe the "alternative states of nature" and ascribe probabilities to these alternative states. In this example, the alternative states of nature are the alternative hypotheses about how many juveniles are supported by a given area of pool habitat. Table 7-3 presents some sample hypotheses and associated probabilities. The probabilities associated with each hypothesis may be generated in a number of ways. One method that can combine multiple types of information is meta-analysis, which pulls together information from multiple sources (Liermann and Hilborn 1997, Myers et al. 2001). Other Bayesian analysis techniques can also be used to combine disparate sources of information. A trademark of Bayesian analysis is the assignment of probabilities to alternative states of nature (Wade 2000). Strengths and weaknesses of the Bayesian approach are described in a special section of Ecological Monographs, edited by Dixon and Ellison (1996). If only limited or ambiguous data are available, expert opinion can be solicited to assign probabilities to the various hypotheses. Numerous texts describe methodologies for soliciting expert opinion (Morgan et al. 1990, Cooke 1991). Readers are encouraged to explore these for full descriptions of the complexity of selecting a group of experts, combining their disparate judgments, and other challenges of this approach. As noted elsewhere in the section, it is impossible to know if expert opinion is correct precisely because we use it in situations for which we have no data. If expert opinion is used to assign probabilities to a set of hypotheses, then the prioritized list that emerges from the decision-analysis process will be a formalization of those opinions.

The next step in setting up a decision table is to associate an “outcome” with each potential action, assuming each of the alternative hypotheses about the state of nature is true. For example, if the hypothesis that pools can support 5 juvenile fish/m^2 is true, then the number of fish expected from the removal of culvert A might be 2,744 fish. In this example the outcome is number of fish, but other appropriate outcome units, such as fish/dollar, may be of interest. This outcome is calculated based on an assessment of the number of pools that would be made available after removal of the culvert. More realistic and detailed decision tables might include additional information such as the number of riffles, the types of pools, the depth of the pools, or the quality of the expected pool habitat. Table 7-4 shows potential outcomes, in total fish, for a number of management actions as a function of fish density in pools.

Finally, we calculate the final expected outcome of each of the potential actions, given the probabilities of the states of nature (Table 7-3). The expected outcome of each action is calculated by summing the expected outcome for each state of nature (Table 7-4) multiplied by the probability that the state of nature is true (Table 7-3). For example, the expected outcome for removal of culvert A is 2744 * 0.1 + 4892 * 0.3 + 5248 * 0.5 + 5786 * 0.1 = 4945. Table 7-5 shows the expected outcome for each of the four potential actions. The largest expected increase in total number of fish is associated with removal of culvert A.

This is an extremely simple example. Hypotheses about the states of nature will often involve more than a single dimension (e.g., more than pool density). Many types of information can be included in the analysis, but there will often only be one or two critical uncertainties that drive a decision. Decision tables provide a structured method for including and communicating uncertainties and can easily be constructed for many of the examples in this document. For example, the methods described in the "Prioritizing Potential Restoration Actions Within Watersheds" section could be modified to include uncertainty about fish response, restoration costs, or habitat quality by using the decision table methodology described here. Another tool for making decisions is a "logic tree", which models the impact of uncertainties in states of nature and in the occurrence of future conditions on possible outcomes (Kessler and McGuire 1999). Logic trees are particularly useful when only subjective probabilities about the states of nature exist.

Using Decision Rules When Empirical Data Are Inadequate

A careful and honest examination of uncertainty in data, predictions, and models will inevitably lead to the identification of situations in which adequate empirical data for making a decision are simply not available. Uncertainty should not lead to inaction. Methods are being developed to allow quantitative analysis of the sensitivity of decisions to uncertainties in the data. For example, sensitivity analyses were used to demonstrate that the best management decision for Hector's dolphin was robust to model uncertainties, and thereby removed uncertainty in the scientific data as an excuse for inaction (Slooten et al. 2000). Where empirical data are inadequate, we strongly discourage basing decisions on biased or imprecise predictions, prioritization systems for which guesswork must be substituted for data, or information that becomes inaccurate or imprecise at the scale for which the decision must apply. Instead, we suggest that managers provide an explicit rationale for the decision that requires minimal data.

The most important characteristics of a decision rule are that it can be documented and that it is robust. Documentation is important because future managers will need to understand the basis for the decision. This requirement prevents arbitrary decisions in the face of inadequate data. Decision rules that are robust to the uncertainties in the information prevent risky management decisions (Schnute and Richards 2001). Decision rules that have been presented in the literature include the following two examples.

The Precautionary Principle can be stated as, “When an activity raises threats of harm to public health or the environment, precautionary measures should be taken even if some cause and effect relationships are not fully established scientifically” (Raffensperger and Tickner 1999). Because this principle shifts the burden of proof to those who create risks and does not define which risks are most important (Hilborn et al. 2001), it has generated much controversy and confusion about its appropriate implementation. However, there are many examples of national and international policies that have been based on the precautionary principle. European environmental law is based on the precautionary principle through the Treaty on European Union, 1992 and international treaties, including the Rio Declaration from the United Nations Conference on Environment and Development, bind the United States to implement the precautionary principle in environmental health protection (Raffensperger and Tickner 1999). While we are not advocating this particular decisionmaking rule, we present it as an example of a relatively simple guiding principle for high-level decisions in the absence of definitive data.

Safe minimum standard (SMS) is another decisionmaking rule that has received considerable attention. The SMS approach is a collective choice process that prescribes protecting a given level of a renewable resource unless the social costs are excessive (Berrens 2001). This approach to making environmental decisions is usually invoked in settings involving considerable uncertainty and potentially irreversible losses. It prioritizes social costs over loss of renewable resources. We present this approach for comparison to emphasize the importance of carefully choosing the decisionmaking principle and documenting exactly what considerations should be involved. The choice of a guiding principle will dictate management decisions until improved information is available.

The choice of a decisionmaking rule need not be purely theoretical. The "An Assessment Approach for Habitat Recovery Planning" section discussed the importance of defining a habitat strategy that includes gathering additional data and taking interim actions. This habitat strategy is an excellent example of how a guiding principle can be used for decisionmaking until adequate data become available. The "Prioritizing Potential Restoration Actions Within Watersheds" section presented guidelines for selecting restoration actions before all of the habitat data are available. Again, this is a simple and effective method for dealing with incomplete information.

Another common approach to formalizing decisionmaking without adequate empirical data or quantitative predictions is a scoring matrix. A scoring matrix can be used to prioritize potential actions, project proposals, potential action sites, or information gathering. The advantage of a scoring matrix is that ranks can be based on weighted priorities, for example project longevity, proximity to other projects, or land ownership. The decision path can be clearly explained and is easily repeatable. As better information becomes available, the matrix can be adjusted. A disadvantage of scoring matrices is that the weights assigned to each priority can dramatically alter the outcome and it is often difficult to specify a satisfactory weighting function in advance. Examples of scoring matrices in current use include the Snake River Salmon Recovery Region Comprehensive Project Scoring Matrix (Snake River Salmon

Recovery Committee 2002), the Lower Columbia Fish Recovery Board Interim Habitat Strategy Project Scoring Sheet (LCFRB Lower Columbia Fish Recovery Board 2001), and the Skagit System Cooperative methodology for rating individual landscape processes (Appendix B). The scoring matrix provided by the Lower Columbia Fish Recovery Board dedicates a section to "Certainty of Success," explicitly including some metrics of uncertainty.

In each of the above examples, it is important to consider whether the decision-strategy is robust to the types of uncertainties that exist. A strategy that would be beneficial under a scenario that has a 50% chance of representing reality but that would be detrimental the rest of the time is not a robust choice. Strategies should be developed so that the outcome is acceptable given the range of possibilities for which there is uncertainty. Again the Hector's dolphin management plan is an excellent example of a strategy that is robust to the uncertainties in the data (Slooten et al. 2000).

Using decisionmaking strategies that require minimal data carries two obligations. First, we must evaluate whether improved information would produce a cost-effective improvement in decisionmaking (value of information analysis). If so, then a strong attempt to reduce uncertainties by gathering more or better information is required. The analyses described in the first example of this section (evaluating a prediction) can be used to identify critical information uncertainties and reduce their impact. Second, we must set a time frame for reevaluating the decision. In the best possible scenario, decision strategies requiring minimal data serve as interim measures until additional information is available.

In conclusion, we emphasize that estimates of uncertainty (quantitative where possible, qualitative for other situations) should be included with all information being considered in a decisionmaking framework. A systematic treatment of uncertainty should include the following elements: a) identification of uncertain events, states of nature, relationships, and parameters; b) determination of the likelihood associated with each potential state or value; c) use of data or models to evaluate consequences of each potential state or value; d) examination of the relationship between uncertain inputs and potential outputs to identify key uncertainties (Mishra 2001). Even where a formal decision theoretic approach is not possible, describing sources and magnitudes of uncertainty is important in providing managers with enough information to weigh potential risks and benefits of possible actions (Rosenberg and Restrepo 1994). A careful examination of the sources and causes of uncertainty will ensure informed decisions and make improvements in both precision and accuracy likely. Quantifications of uncertainty can be formally incorporated into decisionmaking using decision tables. In other situations, simple strategies such as collecting data at multiple scales or incorporating data from other disciplines will provide for more informed decisions. However, a lack of empirical data need not prevent informed decisions from being made in a clear and formal manner. It is possible to implement strategies that require minimal data. Such strategies are preferable to using biased or imprecise predictions, guesswork disguised as data, or information that is inappropriate to the scale of the decision.

Table 7-1. Tools and methods for quantifying and reducing uncertainty.

Class of uncertainty	Brief definition	Habitat example	Methods for quantifying	Possibilities for reducing
Prediction uncertainty	Difference between the modeled response and the true response.	Uncertainty of predicting habitat capacity of a given watershed after instream restoration.	Leave-one-out estimates of prediction error rates. Simulation studies comparing conditions where model was built to those in which it is being applied.	Collect data for conditions in which predictions are required. Do not extrapolate beyond conditions under which model was developed.
Parameter uncertainty	Difference between the true parameter (such as an average or a regression coefficient) and the parameter as estimated from the data.	Uncertainty of parameters describing change in capacity as a function of changes in watershed condition.	Statistical theory for model coefficients derived from data. Sensitivity analysis for model coefficients estimated from other sources.	Collect more data or more accurate data. Collect data over a wider variety of conditions.
Model uncertainty	Difference between natural system and the mathematical equation used to describe it. Includes model form and set of predictors.	Uncertainty in the relationship between habitat conditions and fish capacity. Uncertainty in which habitat descriptors are best predictors of fish capacity.	Statistical descriptions of model fit: AIC, BIC, likelihood ratios, F-statistics.	Consider wide variety of models. Sensitivity analyses.
Measurement uncertainty	Difference between true value and the recorded value.	Uncertainty in measurements of data used to build the predictive model, i.e. fish or redd density under differing habitat conditions.	Test accuracy of measurement technique against standard method or known values.	Improve measurement techniques. Increase the number of replicates. Calibrate biased measurement techniques.
Natural stochastic variation	Inherent random variability.	Natural fluctuations in population size, habitat selection, or habitat conditions.	Variance of the observed data. Variance of the observed data for different sets of conditions.	Collect more replicates for conditions of interest. Stratify data collection.

Table 7-2. Questions to guide the evaluation of predictions.

Prediction uncertainty
How similar are the conditions under which the original information was gathered to those for which the prediction is being made? How sensitive is the model (data, mechanism, and parameter estimates) to site-specific details?

Parameter uncertainty
Is the prediction sensitive to small changes in parameter estimates? If so, how precise are the estimates of those parameters?

Model uncertainty
What are the assumptions on which the prediction is based? How sensitive is the prediction to these assumptions?

Measurement uncertainty
Could any of the information on which the prediction is based be biased? How precise and how accurate are the data?

Natural Variability
Can measurements be stratified across conditions to reduce the effects of natural variability?

Table 7-3. Example alternative hypotheses about the states of nature (i.e. density of fish per m^2 of pool habitat). The probabilities describe the relative likelihood that each hypothesis is true. All probabilities must sum to one.

Hypothesized fish density	5	10	15	20
Probability hypothesis is correct	0.1	0.3	0.5	0.1

Table 7-4. Expected outcomes for potential habitat actions (total fish) as a function of hypothesized fish density per pool.

Potential action	Hypothesized fish density per pool			
	5	10	15	20
Remove culvert A	2744	4892	5248	5786
Remove culvert B	2844	3400	3858	6457
Remove riprap	2012	4172	4260	4340
Add wood	1568	3410	5963	6230

Table 7-5. Overall expected outcomes (increase in total number of fish) of potential actions.

Potential action	Expected outcomes
Remove culvert A	4945
Remove culvert B	3879
Remove riprap	4017
Add wood	4784

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GLOSSARY

Anadromous

Moving from the sea to fresh water for reproduction.

Anthropogenic

Caused or produced by human action.

B-IBI

Benthic Index of Biological Integrity

Biodiversity

Biological diversity in an environment as indicated by numbers of different species of plants and animals.

Biological integrity

Biota

The flora and fauna of a region.

Channel width (wetted width and bankfull width)

Bankfull width is the channel width between the tops of the most pronounced banks on either side of a stream.

Culvert

Buried pipe or covered structure that allows a watercourse to pass under a road or underground.

CWA

Clean Water Act

Distributary (also, distributary channel)

A branch of a river or stream that flows away from the main channel and does not rejoin it.

Disturbance

Introduction of an unwanted condition into a system or interference with a habitat's normal or existing conditions.

Ecoregion

An area determined by similar land surface form, potential natural vegetation, land use, and soil; it may contain few or many geological districts (Omernik 1986).

Ecosystem

The system of all the living and dead organisms in an area and the physical and climatic features that are interrelated in the transfer of energy and material.

EIA

Effective impervious area

Endangered species

A species in danger of extinction throughout all or a significant portion of its range.

ESA

U.S. Endangered Species Act

ESU

Evolutionarily significant unit**:** An ESU is "a population or group of populations that are 1) substantially reproductively isolated from other populations, and 2) contribute substantially to the ecological or genetic diversity of the biological species (Myers et al. 1998)." It is sometimes represented as a spatial area as well.

Field data (compare to remotely sensed data)

Floodplain

Lowland area bordering a stream, onto which it spreads at flood stage.

GIS

Geographical information system

Glide

Relatively slow and shallow stream section with water velocities of 10-20 cm/s and little or no surface turbulence.

Habitat enhancement

Improving habitat conditions from an existing state.

Habitat restoration

Restoring habitat conditions to some predisturbed state (Koski 1992 [in citations?]).

Habitat unit

Relatively homogenous area of the stream channel that differs from adjoining areas in dept, velocity, and substrate characteristics (Armantrout 1998).

Hydromodification

IBI

Index of biological integrity

Impoundment

Interim strategies and actions

Landsat

Landsat satellites supply global land surface images.

LWD

Large woody debris: A large piece of woody material, such as a log or stump, that intrudes into a stream channel [specify diameter and length greater than certain amounts?].

Main stem

Principle stream or channel of a watershed system.

Multi-species management

Outlier

In statistics, any data point exhibiting anomalous behavior.

Parr

A young salmonid actively feeding in freshwater.

Population

A group of individuals of a species living in a certain area that maintain some degree of reproductive isolation. [Alt (Issues of Scale section): Collection of individuals making up a gene pool that has a continuity in time due to the reproductive activities within the population (MacLean and Evans 1981)]

Pool:riffle:glide ratio
Ratio of the respective surface areas or lengths of pools to riffles to glides in a given stream reach, often expressed as the relative percentage of each category.

Reach
A section of stream between two points.

Recovery

Redd
Nest in gravel, dug by a fish for egg deposition, and associated gravel mounds.

Refugia

Remotely sensed data (compare to field data)
Data gathered at a remote station or platform, as in satellite or aerial photography.

Restoration

Riffle
Shallow section of a river or stream with rapid current and a surface broken by boulders, rubble, or gravel.

Riparian
Relating to or situated on the area between a stream or other body of water and the adjacent upland.

Riprap
Layer of large, durable materials used to protect a stream bank from erosion; may also refer to the materials used, such as rocks or broken concrete.

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Salmonid

Fish of the family *Salmonidae*, including salmon, trout, and chars.

Seral

Of, relating to, or constituting an ecological sere. (A sere is a series of ecological communities formed in ecological succession.)

Side channel

Smolt

Juvenile salmonid in its seaward migrant stage.

SMS

Safe minimum standard

Species

Interbreeding populations. [Ck other defs, inclu. biological vs legal.]

Stochastic

Of or relating to uncertainties or random variables.

SWAM

Salmonid Watershed Analysis Model: a large scale landscape analysis for identifying high priority areas for salmon habitat restoration.

Taxa

Plural of taxon, a taxonomic group or entity.

TRT

Technical Recovery Team: The TRT is responsible for establishing biologically based ESA recovery goals for listed salmonids within a given recovery domain. Members serve as science advisors to the recovery planning phase.

Threatened species

A species not presently in danger of extinction but likely to become so in the foreseeable future.

VSP

Viable Salmonid Population

WDFW
Washington Department of Fish and Wildlife

Watershed (also drainage basin or catchment area)
The land drainage area of a stream.

APPENDIX A. ESTIMATING CHINOOK SPAWNER CAPACITY FOR THE STILLAGUAMISH RIVER

Lisa Holsinger

One important task of TRTs is evaluating both current and historical capacity of the habitat to support salmonids as juveniles and adults. These estimates are needed to inform and assess the population viability analyses, conducted for developing recovery goals. Current and historical estimates of capacity can provide a context in which to evaluate how reasonable the results of our viability estimates are. For example, if viability estimates indicate population abundance of 5,000 for a population to be viable, estimates of spawner capacity can indicate whether 5,000 fish is within the bounds of what the watershed may have historically supported. Our methods are generally based on techniques used to evaluate coho production in the Skagit and Stillaguamish rivers (Beechie et al. 1994; Pess et al. 1999).

Riverine habitats function at different spatial and temporal scales, ranging from the watershed level down to microhabitats (Frissel et al. (1986). Larger scale systems (such as watershed and segment) generally operate over a 100 to 1000 m linear spatial area and persist for 1,000 to 1,000,000 years (Frissell et al. 1986). Extrinsic forces driving these larger systems range from glaciation, volcanism, tectonic uplift, and climatic shifts to earthquakes, very large landslides, and alluvial or colluvial valley infilling. Reach-scale systems operate at intermediate scales of about 100 m and 10-100 years, and are driven by events such as debris torrents, landslides, and log input or washout. Smaller scale, ‘habitat-unit’ systems are typically controlled by events or processes occurring at spatial scales of 1 to 100 m and over shorter time periods (persistence of 1 to 10 years) (Frissel et al. 1986). Habitats at this level include pools and riffles as well as glides, rapids, side-channels and backwater pools, and have characteristic bed topography, water surface slope, depth, and velocity patterns (Bisson et al. 1982). These habitat units are driven by input or washout of wood, boulders, etc., small bank failures, flood scour or deposition, thalweg shifts, and numerous human activities (Frissel et al. 1986).

As a case study, we estimated the historical and current capacity of the Stillaguamish River for adult chinook salmon with a focus on the habitat-unit scale (e.g., pool, riffle, and glide). The habitat-unit scale is critical to the development estimates of salmonid utilization because it serves as a basis for extrapolating to larger scales such as the stream-reach (e.g., 0.5 kilometers to several kilometers) and watershed scale. For example, relationships between fish abundance and channel characteristics at the habitat-unit scale can be extrapolated to larger scales, such as the stream-reach, to help identify the potential magnitude of differences in spawning populations (Montgomery et al. 1999). This analysis of capacity allows us to estimate the maximum number of adult chinook that the Stillaguamish River produced historically, as well as the system's potential for production today.

Methods

Our approach for estimating capacity essentially has two steps. One, we assess the amount of habitat available for chinook spawning, and two, we associate fish numbers to that habitat. To assess available habitat, we start at the watershed scale and ask where adult chinook distribute themselves over the basin for spawning. Within that distribution, we describe spawning habitat at the reach scale by identifying streams either as large main stems or as small main stems and tributaries. We quantify the amount of each reach-scale habitat by estimating the total stream length for small main stems and tributaries, and the total stream area for large main stems. For small main stems and tributaries, habitat can only be described at the reach scale, and not at the finer-unit scale, due to a lack of habitat-unit data. However, large main stems are further characterized by estimating the proportion of reach-scale habitat that is pool, riffle, glide, etc., and by how much of each habitat-unit is suitable for spawning.

In the second step, we assign fish numbers to habitats with different methods depending on the reach-scale type of habitat. For small main stems and tributaries, we estimate the total number of redds possible by multiplying stream length by an estimate for the expected number of redds per kilometer. For large main stems, we calculate how many redds would fit into our estimated stream area, by assuming an estimate for redd area. Then we convert estimates of total redds for each reach-scale habitat type to total adults using an estimate for number of adults per redd.

Watershed-scale Chinook Spawning Distribution

Streams were identified as accessible to chinook salmon based on the location of barriers and stream gradient. The historical distribution of chinook includes areas below natural barriers (i.e. waterfalls, cascades) and with low-gradient streams (less than 4% gradient) (Montgomery et al. 1999). Similarly, current fish distribution includes areas below anthropogenic barriers (primarily culverts) and natural barriers, and with low-gradient streams. At the upstream end, Granite Falls in the South Fork Stillaguamish River was not included in either current or historical capacity estimates. Historically, few chinook were able to ascend Granite Falls. At present, a fish ladder allows chinook to pass upstream, but observed productivity is low (Kit Rawson, Tulalip Tribes, pers. commun.). We defined the downstream limit of chinook spawning as the upper extent of tidal influence, which occurs at about the confluence of Cook Slough and Stillaguamish River, and Pilchuck Creek with the Stillaguamish River (Figure A-1).

Figure C-1. Location of Stillaguamish River basin and map of major rivers.

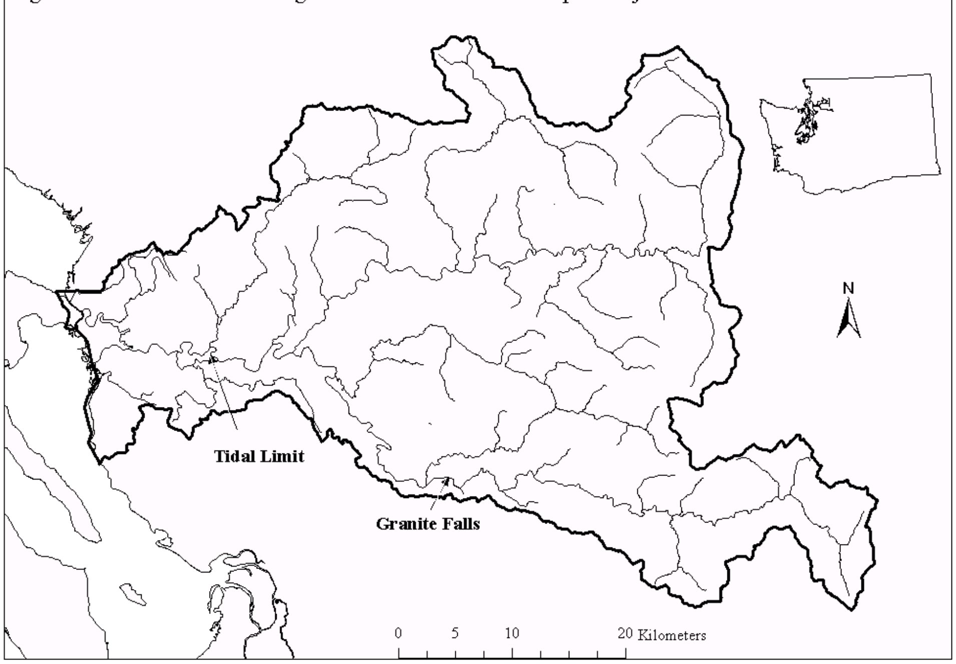

Figure A-1. Location of Stillaguamish River Basin and map of major rivers.

Reach Scale Chinook Spawning Habitat

The main spawning areas of chinook salmon are in larger tributary streams and main stems of rivers (Miller and Brannon 1981). Chinook use of these freshwater habitats varies depending on the size of bankfull channel widths. Chinook redd density in the Stillaguamish River generally increases up to about 25 m bankfull channel width, and then sharply declines (Figure A-2) (Montgomery et al. 1999). Similarly, channel morphology varies at different scales as a factor of channel width and large woody debris (LWD) loading. Pool frequency increases with bank-full channel width and LWD loading up to about 20-25 meter channel width, and then drops off (Montgomery et al. 1995). Pools in these smaller streams can be formed by individual pieces of wood, which cross streams and form stable obstructions (Abbe 2000). In larger streams, channel morphology is most significantly affected by large logjams, which form stable obstructions that create pools but at a much lower frequency (Abbe 2000). Salmon usually spawn at the transition between pools to riffles, or in areas associated with a lateral bar deposition (Bjornn and Reiser 1991).

Because chinook use of streams and channel morphology types vary with channel width, we classified streams to reach-scale habitat types by bankfull width. Small streams (or mainstem/tributary habitat types) are ≤25 m bankfull channel width, and large mainstem habitats are >25m width. Additionally, streams less than 5 m bankfull width were regarded as too small for consistent chinook spawner production based on data from the Stillaguamish, Snohomish, Skagit, Suiattle and White Rivers (Don Hendrick, WDFW, personal communication; James Doyle, USFS, personal communication). Chinook are occasionally observed spawning in smaller streams (D. Hendrick, pers. commun., Vronskiy 1972); however, we wanted to include stream habitat where the majority of chinook consistently spawn.

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Figure C-2. Comparison of chinook redds per kilometer to bankfull width measurements, Skagit River (based upon data in Montgomery et al. 1999)

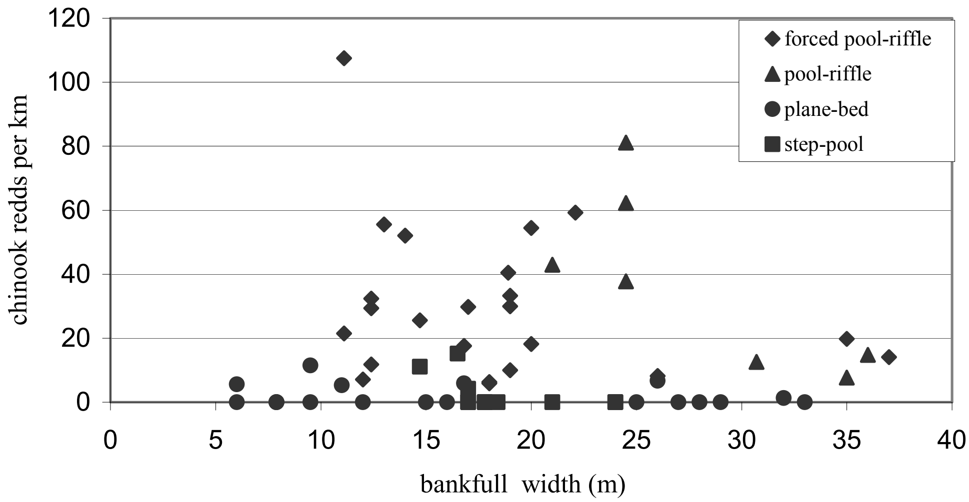

Figure A-2. Comparison of chinook redds per kilometer to bankfull width measurements, Skagit River (based on data in Montgomery et al. 1999).

We estimated bankfull channel width for both historical and current stream conditions using regression models. For historical conditions, we developed a regression to predict channel width using channel width measurements from 1860s General Land Office Survey notes (B. Collins, unpubl. data, Beechie et al. in review) and basin drainage area for the Stillaguamish River:

Historical channel width = $10^{[-2.4 + 0.54 * \log(\text{drainage area})]}$, Adjusted $R^2 = 0.70$

Basin drainage area was derived using 7.5 minute (30-meter) digital elevation model (DEM) data. For current conditions, the regression model was also developed using channel width data and basin drainage area:

Current channel width = $10^{[-1.5 + 0.43 * \log(\text{drainage Area})]}$, Adjusted $R^2 = 0.68$

Channel widths were measured primarily from aerial photographs, but included field measurements as well.

Using these regression relationships, we developed a GIS-data layer of channel width for the entire Stillaguamish Watershed based on the DEM-derived drainage area data. The channel width predictions were then applied to categorize all streams in a 1:24,000 scale hydrography into the 3 categories: 1) <5m bankfull channel width, 2) 5 to 25 m width; and 3) >25 m width.

Chinook Capacity in Small Streams

Habitat in small streams can only be described at the reach-scale due to a lack of data describing the proportions of pools, riffles and other unit-scale habitats. Likewise, the data available for estimating chinook use of small streams (5 to 25 m width) is expressed at the reach scale, specifically, as number of redds per kilometer of stream length (Montgomery et al. 1999).

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Hence we estimated chinook spawner capacity in small streams as a function of stream length, number of redds per kilometer, and number of adults per redd:

$$N_{adults} = [(\text{Total stream length} * \text{Redds/km})] * \text{No. Adults/redd}$$

In generating our historical capacity estimate, we calculated the total length of small streams within historical chinook spawning areas from the 1:24,000 hydrography data. Additionally, we included the total length of small, non-pond like side-channels estimated for historical conditions (Pess et al. 1999). For the current capacity estimate, we excluded those streams and side-channels that were blocked (by culverts or dams) or otherwise isolated (Table A-1).

Table A-1. Estimates of stream length (m) of small streams (5 to 25 m bankfull width) under current and historical conditions for the Stillaguamish River.

	Habitat type	North Fork	South Fork	Main stem
Historical conditions	Small streams and tributaries	209,348	87,148	39,572
	Non-pond-like side channels	10,939	2,667	28,007
Current conditions	Small streams and tributaries	207,173	93,198	45,291
	Non-pond-like side channels	4,462	0	11,079

For estimates of redds per kilometer, we assumed that historically, channel morphology in small streams was determined primarily by large woody debris (LWD) loading. Such in-channel obstructions produce forced pool-riffle habitats -- channels in which the majority of pool and bar forms are forced by flow convergence, divergence, and turbulent scour associated with obstructions (Montgomery et al. 1995). Forced-pool riffle habitats, formed by woody debris, may be considered indicative of undisturbed conditions (Lunetta et al. 1997). Therefore, data for number of redds per kilometer only include counts from forced pool-riffle habitats (versus pool-riffle and plane-bed) for historical conditions (Montgomery et al. 1999). Redd density data were collected from 50 reach-level surveys in five tributaries to the Skagit between 1991 and 1996. The median value for redds per kilometer in forced pool-riffle channels was 29.6, and the 10th and 90th percentiles of the range are 7.0 and 57.5 redds per kilometer, respectively (Table A-2).

Table A-2. Values used to vary biological parameters for current and historical capacity estimates. Redd size data collected in the North Fork Stillaguamish; number of adults per redd from the Snohomish and Stillaguamish (D. Hendricks, unpubl. data); number of redds per kilometer from the Stillaguamish and Skagit rivers (Montgomery et al. 1999).

	Redd size (m^2)	No. adults per redd	No. redds per kilometer	
			Historical	*Current*
90th percentile	4.9	3.5	57.5	13.8
Median	14.1	1.9	29.6	2.5
10th percentile	27.9	1.4	7.0	0.0

Spawner survey data are also available describing chinook use specific to the North Fork Stillaguamish River from nine river reaches (Table A-3) (Montgomery et al. 1999). However, these data were collected over longer time spans (up to 23 years), and with less certainty that the identified habitats (in particular, forced pool-riffle) persisted over the time period that chinook were observed. These data were useful for our current capacity estimate where redd counts were needed across a range of habitat types, which could vary across time and space (Table A-3). These average counts were first applied to the tributaries from which data were collected (i.e. Boulder River, Squire Creek, etc.). For the remaining small streams, we summarized the data from the nine streams (Table A-3), and applied the median (2.5 redds/km), 10th (0 redds/km) and 90th (13.8 redds/km) percentiles of the range in the current capacity estimate (Table A-2).

We estimated the number of adults per redd using data that describes the number of males per female from: 1) carcass recovery survey data from North Fork Stillaguamish, Skagit and Snohomish Rivers, 2) Sunset Falls counts, Snohomish River, 3) broodstock collection data from North Fork Stillaguamish and Skagit Rivers, 4) mark/recapture study from the North Fork Stillaguamish, and 5) Gill drift net test fishery data from the Skagit River (Hahn et al. 2001, Don Hendrick, unpubl. data; WDF 1976). The ratio of males:females plus one is equivalent to the number of adults per redd. The median value for number of adults per redd was 1.9, and the 10th and 90th percentiles of the range were 1.4 and 3.5, respectively.

To illustrate the full range of potential historic and current capacity values, we calculated capacity estimates using the 10th percentile, median and 90th percentile ranges of all spawner biological variable values (redds per km and adults per redd). Stream length included only one measurement each for historical and current conditions. All capacity calculations include estimates for the North Fork Stillaguamish chinook population, and the South Fork/mainstem Stillaguamish population.

Table A-3. Data on number of chinook redds for repeated spawner surveys along the North Fork Stillaguamish (Montgomery et al. 1999).

Location	Years of data	Channel type	Average redd/km
Boulder River	16	Pool-riffle	4.2
Squire Creek	23	Pool-riffle	6.4
Furland Creek	2	Forced pool-riffle	13.8
Ashton Creek	2	Forced pool-riffle	8.8
Browns Creek	10	Forced pool-riffle	2.5
Brooks Creek	4	Plane-bed	0.0
Rollins Creek	6	Plane-bed	0.0
Dicks Creek	4	Plane-bed	0.0
Segelson Creek	2	Plane-bed	0.0

Historical Chinook Capacity in Large Streams

We estimated historical chinook spawner capacity in large streams (>25 m bankfull width) by determining the amount of area in a river that has habitat suitable for spawning, and calculating the number of redds that fit in that area. Then we estimated the number of adults as a function of number of fish per redd. The equation for historical capacity in large streams is as follows:

$$N_{adults} = [\ (\text{Stream Area}) * (\text{Percent area suitable for spawning})\ ] * \text{No. adults/redd}$$

To calculate stream area in the Stillaguamish Basin, we first generated an estimate of wetted widths across all stream reaches for the time period when chinook spawn (mainly August/September). We measured widths from 1:24,000 digital orthophotos of the Stillaguamish Watershed. The photos were taken in mid- to late-July 1998. Mean monthly flows in July are 25% to 45% higher than in September (generally, the peak of spawning), which means we will overestimate spawner abundance by some fraction. However, available spawning area does not increase in direct proportion to stream flow (i.e. 25% more flow does not equal 25% more spawning area), so overestimation may not be significant. We then developed a regression model to predict wetted widths based on cumulative stream lengths (total stream length above each wetted width measurement) and stream order data, derived from 30-metre DEMs:

$$\text{Wetted width} = 10\text{^}[-2.59 + 0.56\log(\text{Cumulative flow length}) + 0.36(\text{Stream order})]$$

$$\text{Adjusted } R^2 = 0.77$$

With this regression, we developed a GIS-data layer of wetted width for the Stillaguamish Basin, and associated the wetted width predictions to stream reaches in the hydrography layer. We then calculated stream area for each reach as a function of wetted width and stream length. These stream area estimates were used for calculations of both historical and current capacity (Table A-4). In addition, we included estimates of stream area for non-pond like side-channels for historical conditions (Pess et al. 1999). All of these side channels are presently isolated from streams accessible to anadromous fish.

Table A-4. Estimates of stream area (m^2) under historical and current conditions for the Stillaguamish River

Time period	Habitat type	North Fork	South Fork	Main stem
Historical conditions	Large streams	2,616,542	1,842,123	1,318,877
	Non-pond-like side channels	70,676	0	98,106
	Total	2,687,218	1,842,123	2,416,983
Current conditions	Large streams	2,532,976	1,696,890	1,023,863
	Non-pond-like side channels	0	0	0
	Total	2,532,976	1,696,890	1,023,863

To describe the stream area suitable for spawning under historical conditions, we conducted field surveys in western Washington streams that are in relatively undisturbed condition (North Fork Sauk River, mainstem Sauk River, South Fork Stillaguamish River, Squire Creek, and the South Fork Hoh River). Pess and Abbe (1994) developed criteria for measuring the area suitable for spawning by describing habitat characteristics in the North Fork Stillaguamish River where redds were observed. Geist et al. (2000) similarly used characteristics of spawning habitat measured within local spawning areas to evaluate chinook salmon habitat suitability in the Columbia River. In the Stillaguamish River, substrate averaged 74 mm (range of 45 – 120 mm) in width, and was typically composed of large gravel and small cobble (Pess and Abbe 1994). Depths averaged 0.5 m (range of 0.3 – 1.5 m), and velocities averaged 2 feet per second (range of 1 – 3 feet per second). Healey (1991) summarized comparable values from the literature for water depth and velocity in chinook spawning beds.

In our habitat surveys of reference sites, the average values for substrate and depth were the primary variables used to estimate suitable spawning area, in addition to channel bed morphology. Channel bed morphology indicates the location of sub-surface flow (such as at the junction of a pool's tailout and the head of a riffle), which is important in redd site choice by chinook (Vronskiy 1972, Chapman 1943, Russell et al. 1983). Once we identified the location of sub-surface flow (i.e. the primary spawning site), we measured the area suitable for spawning within a habitat (riffle, pool, glide) by surveying the extent of appropriate average substrate and depth values. We calculated historical capacity estimates where percent habitat spawnable was segregated by habitat type (Table A-5). However, we had less certainty that habitat composition was reflective of historical conditions because of the limited size of our reference sites. Therefore, we summarized the percent of area suitable for spawning ("percent habitat spawnable") across all habitat units combined (Table A-5).

For the capacity estimate, we assumed that redds are uniformly distributed and positioned immediately adjacent to one another, without a larger territorial boundary. Geist et al. (2000) found that when redd densities were near capacity, clusters or redds were uniformly distributed and inter-redd distances ranged from 2 to 5 m. In the Stillaguamish River, redds were estimated to have a median size of 14.1 m^2 (range of 4.9 m^2 and 27.9 m^2 as the 90th and 10th percentiles, respectively) (Table A-2). These redd sizes compare well with redd areas reported by others (Table A-6) (Healey 1991). If we assume redds have an approximately circular shape and thus, radius of 2.11 m, distances between redds in the Stillaguamish River would fall roughly within the range observed by Geist et al. (2000).

We calculated a range of spawner capacity estimates using the 10th percentile, median and 90th percentile of values for all variables (redd size, number of adults/redd, percent habitat spawnable) (Tables A-2 and A-5), except stream area for which we only had one measurement (Table A-4). Our best estimate for historical capacity included percent habitat spawnable with habitat-units combined, but we also calculated capacity estimates with percent spawnable evaluated separately by habitat type.

Table A-5. Percent habitat composition values, and values used to vary percent of habitat suitable for spawning for historical and current conditions. For current conditions, percent of habitat with observed spawning was calculated using WDFW redd data from 1998, 1999 and 2000; and percent of habitat suitable for spawning was estimated using field measurements of habitat (see text).

Data source	Parameter	Riffle	Glide	Pool	Rapid	Backwater pool	Habitat units combined
Historical conditions							
Reference site data	Percent habitat composition	26.0	25.8	47.0	--*	--*	100.00
	Percent habitat spawnable						
	90th percentile	20.0	18.0	3.0			9.50
	Median	17.0	1.0	2.0	--	--	3.00
	10th percentile	3.0	0.0	0.0			0.00
Current conditions							
WDFW habitat survey data	Percent habitat composition	46.0	39.0	11.0	3.0	1	100.00
WDFW redd survey data	Percent of habitat spawned						
	90th percentile	3.4	3.4	3.3	3.8	0	5.50
	Median	1.8	0.2	0.0	0.4	0	0.38
	10th percentile	0.0	0.0	0.0	0.0	0	0.00
NMFS habitat survey data	Percent habitat spawnable						
	90th percentile	77.1	50.2	0.0	0.0	0	50.00
	Median	18.8	12.0	0.0	0.0	0	2.70
	10th percentile	1.0	0.0	0.0	0.0	0	0.00

* rapid and backwater pool habitats were not surveyed

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Table A-6. Summary of published information of redd size (Healey 1991).

Source	Type*	Redd area (m^2) Range	Mean
Chapman (1943)		2.4 - 4.0	
Burner (1951)	O	4.8 - 6.5	
	S	2.4 -4.1	
Vronskiy (1972)	S	4.0 - 15.0	
Nelson & Banford (1983)	S	0.5 - 27.5	9.5
Chapman et al. (1986)	O	2.1 - 44.8	17.0

*S = stream-type (spring-run) chinook, O = ocean-type (fall and summer) chinook

Current Chinook Capacity in Large Streams

To estimate current capacity in large streams, we used field data for habitat distribution and redd location (L. Lowe, Washington Department of Fish and Wildlife, unpubl. data) and field data characterizing habitat suitable for spawning from river miles 15 to 30 of the North Fork Stillaguamish River (NMFS, unpubl. data). We digitized WDFW field maps describing habitat units (pool, riffle, glide, etc.) into a Geographic Information System (GIS) database, and summarized the habitat composition across the 15-mile reach (Table A-5). We also digitized redd locations for the same 15 mile stretch from 1998, 1999 and 2000 field surveys.

We estimated current capacity similar to methods for historical capacity, except we first described the habitat composition of the river, and then estimated the percent of habitat suitable for spawning based on those habitat units:

$$N_{adults} = \frac{[(SA * \%Spwn)_{pools} + (SA * \%Spwn)_{riffles} + (SA * \%Spwn)_{glide}]}{\text{Redd size}} * \text{No. Adults/redd}$$

%Spwn = Percent of habitat spawned or suitable for spawning
SA = Stream Area

We estimated the percent of habitat that is suitable for spawning in 2 ways. First, we estimated the actual percent of habitat with observed spawning using WDFW redd survey data. We used GIS to overlay the combined 3 years of redd survey data with the habitat-unit maps. A buffer was generated around all redd locations, to represent area spawned. We assumed a radius of 2.11 m, which would be equivalent to a 14 m^2 redd size. For each habitat type, the percent of habitat spawned was calculated as the area within buffers divided by the area of the habitat type (WDFW redd survey data in Table A-5). Second, we estimated the percent of habitat suitable for spawning using our field survey data where spawning habitat was characterized by substrate, velocity, depth and channel bed morphology, as described for historical large stream capacity (habitat survey data in Table A-5). Both estimates of percent habitat spawned and percent habitat spawnable were calculated by habitat type and for all habitat units combined.

Similar to historical estimates for large streams, we calculated current spawner capacities in large streams using the range of values for all variables (redd size, number of adults per redd, percent habitat spawnable by habitat type) (Tables A-2 and A-6), except stream area where only one measurement was possible (Table A-4). Our habitat composition estimates are robust given the substantial length of stream area surveyed. Therefore, we consider current capacities with percent spawnable measured by habitat type to be the best estimates. For comparison, we also included capacities where percent spawnable was measured across all habitat-units combined.

Sensitivity Analyses

We evaluated the sensitivity of our estimates to parameter values used in the capacity equation, as demonstrated by Gray and Megahan (1981). We conducted the sensitivity analysis on the historical capacity estimate in the North Fork Stillaguamish River. We selected the range of values as the 10th and 90th percentile of each variable (Tables A-2 and A-5). The base level of capacity was computed using median values for all variables. We then changed each input variable across its lower and upper range of values while holding other variables constant. Finally, the results were plotted for capacity as a relative percentage of change due to variation in each variable (Figures A-3 and A-4).

Results

Spawner Capacity Estimates

Capacity estimates based on 90th percentile of all input variables are as much as three orders of magnitude larger than estimates using the 10th percentiles for both the North Fork Stillaguamish chinook population and the South Fork/mainstem population, under historical conditions as well as current (Table A-7). The most realistic capacity estimates are probably those determined using the median values for the biological parameters. It is less clear which statistic is appropriate for estimating percent habitat spawnable under both historical and current conditions. If we assume that reference site data underestimate available habitat under historical conditions, we would use the 90th percentile for percent spawnable for historical capacity. This may be a reasonable assumption, given that some of the reference sites (e.g. Upper Sauk and

South Fork Stillaguamish Rivers) are not likely as pristine as we expect streams were under historical conditions. For current conditions, we would use the 90th percentile for percent habitat spawnable with the WDFW redd survey data if we assume that the North Fork Stillaguamish River is presently underutilized. This assumption may also be fair given that overall, higher proportions of the habitat types (in particular, riffles and glides) were characterized as suitable for spawning than were actually observed spawned in by chinook (Table A-5). Finally, results from NMFS habitat survey data seem high and perhaps overestimate capacity by failing to capture factors other than substrate, depth, and flow which may be important for spawning site selection. For NMFS habitat data, the median value for percent spawnable may be more appropriate for capacity estimates.

Using the above assumptions, the best estimate of historical spawner capacity for the North Fork Stillaguamish chinook population is about 46,800 adults. Current capacity would be in the range of 24,100 to 47,000 adults. For the South Fork/mainstem population, historical capacity would occur at about 50,600 adults, and current capacity would range between 25,000 and 49,700 adults.

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Table A-7. Range of adult capacity estimates (number of spawners) for the North Fork Stillaguamish chinook population, and the South Fork/mainstem Stillaguamish chinook population, where the combined biological parameter values are varied and percent habitat suitable for spawning is varied. Estimates in bold represent those we believe are the most likely values (see text)

Location	Percent habitat spawnable	*90th percentile*	*Median*	*10th percentile*	Comments
North Fork	*Historical conditions*				
Stillaguamish	Reference site data				Percent habitat
	90th percentile	228,389	**46,834**	14,465	spawnable calculated
	Median	102,484	23,275	5,996	independent of habitat
	10th percentile	45,342	12,582	2,152	type.
	Current conditions				
	WDFW data				
	90th percentile	129,513	**24,141**	8,874	Percent habitat
	Median	25,166	4,615	1,855	spawnable calculated
	10th percentile	8,533	1,503	736	by habitat type
	NMFS data				
	90th percentile	1,013,582	189,569	68,340	Percent habitat
	Median	252,106	**47,080**	17,120	spawnable calculated
	10th percentile	16,964	3,080	1,303	by habitat type
South Fork	*Historical conditions*				
and mainstem	Reference site data				Percent habitat
Stillaguamish	90th percentile	254,882	**50,622**	16,503	spawnable calculated
	Median	102,182	22,048	6,232	independent of habitat
	10th percentile	32,879	9,080	1,570	type.
	Current conditions				
	WDFW data				
	90th percentile	137,179	**25,027**	8,741	Percent habitat
	Median	25,096	4,054	1,202	spawnable calculated
	10th percentile	7,231	711	0	by habitat type
	NMFS data				
	90th percentile	1,086,786	202,720	72,616	Percent habitat
	Median	268,860	**49,668**	17,598	spawnable calculated
	10th percentile	16,286	2,406	609	by habitat type

Sensitivity analyses—In small streams of the North Fork Stillaguamish River, historical capacity was most sensitive to changes in redds per kilometer, and secondly, adults per redd (Figure A-3). Historical capacity in large streams showed the greatest sensitivity to percent habitat spawnable, followed by redd size (Figure A-4). This estimate was also moderately sensitive to changes adults per redd (Figure A-4).

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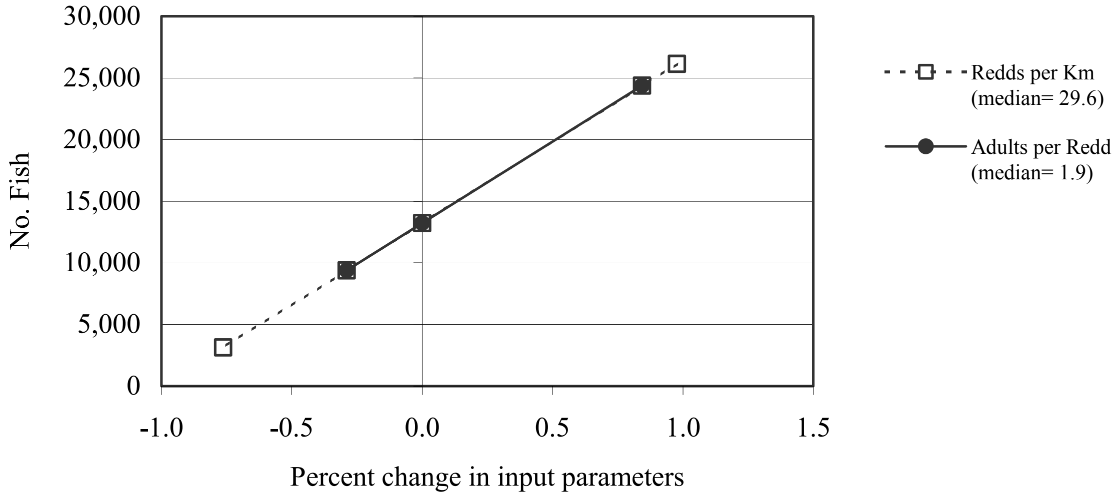

Figure A-3. Range of historical capacity estimates observed by varying parameters in the capacity equation for small streams (≤25 m bankfull width) in the North Fork Stillaguamish River.

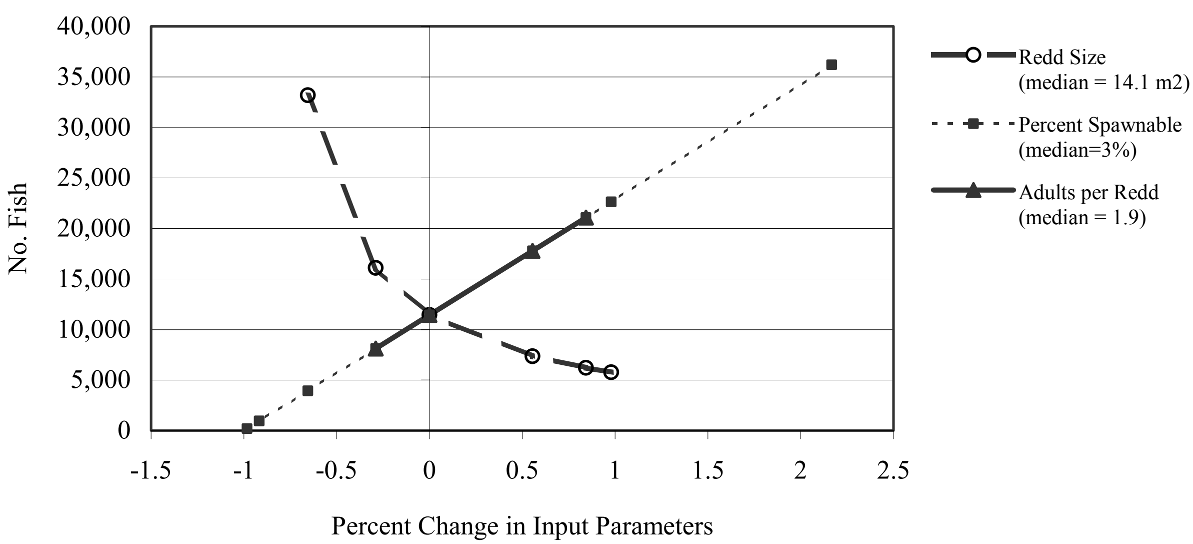

Figure A-4. Range of historical capacity estimates observed by varying parameters in the capacity equation for large streams (>25 m bankfull width) in the North Fork Stillaguamish River.

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APPENDIX B. RESTORATION OF HABITAT-FORMING PROCESSES: AN APPLIED RESTORATION STRATEGY FOR THE SKAGIT RIVER

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Escapement levels of Pacific Northwest and British Columbia salmon stocks have declined dramatically in the past century due to habitat loss, high levels of harvest, and changes in ocean conditions. Land-use induced freshwater habitat losses were associated with the decline of nearly all of the stocks at-risk in a recent study by Nehlsen et al. (1992). However, the recognition of the causes of these declines and the desire to restore salmon runs has not led to specific plans for recouping habitat losses in large watersheds. Rather, most habitat restoration actions have been conducted in an unplanned and uncoordinated fashion.

In 1997, the Skagit Watershed Council (SWC) was formed to support the voluntary restoration and protection of salmon habitats in the Skagit River basin of Washington State. Today the Council is comprised of 36 member organizations including private industrial and agricultural interests, state and federal agencies, local governments, tribes, and environmental groups. In 1998, the Council adopted a salmon habitat protection and restoration strategy that recognizes the influence of land use and resource management activities on natural landscape processes, which result in changed habitat conditions (Skagit Watershed Council 1998). Since 1998, members of the SWC have completed an interim application of the strategy, which identifies causes of degraded habitats in the watershed and restoration actions that are needed to restore habitats over the long term. In this paper we briefly describe the strategy and methods, and present the preliminary findings of the analyses. We also discuss costs of the analysis relative to projected costs of restoration actions.

Overview of the Restoration Strategy

Figure B-1 is a conceptual diagram illustrating how watershed controls (ultimate and proximate) and natural landscape processes combine to form various habitat conditions. Ultimate controls are independent of land management over the long term (centuries to millennia), act over large areas (>1 km^2), and shape the range of possible habitat conditions in a watershed (Naiman et al. 1992, Beechie and Bolton 1999). Proximate controls are affected by land management over the short term (≤ decades), act over smaller areas than independent controls, and are partly a function of independent factors (Naiman et al. 1992). Landscape processes are typically measured as rates and characterize what ecosystems or components of ecosystems do. For example, sediment or hydrologic processes in a watershed may be characterized by the rates (volume/area/time period) at which sediment or water is supplied to and transported through specific locations of a watershed. Some riparian related functions can be viewed similarly. For example, large woody debris (LWD) "recruitment" is synonymous with the idea of supply while LWD "depletion" is the result of both LWD transport and decay rates. Natural rates of landscape processes are here defined as those that existed prior to widespread timber harvest, agriculture, or urban development.

The SWC's habitat protection and restoration strategy describes a scientific framework and set of procedures for identifying and prioritizing activities that restore or protect aquatic habitat (Skagit Watershed Council 1998). The scientific framework strives to identify: 1) the natural landscape processes active in a watershed, 2) the effects of land use on natural processes, and 3) the causal relationships between land use and habitat conditions. It focuses not on the symptoms of watershed degradation but on the fundamental causes, and encourages restoration and protection of natural landscape processes that formed and sustained the habitats to which salmon stocks are adapted. Justification of this approach is based on our understanding from current literature that natural landscape processes create and maintain the "natural" habitat conditions in which native aquatic and riparian species have adapted (e.g., Peterson et al. 1992, Doppelt et al. 1993, Reeves et al. 1995, Ward and Stanford 1995, Beechie et al. 1996, Kauffman et al. 1997).

We apply the strategy by systematically identifying land-use disruptions to landscape processes that form salmon habitat. These processes include peak flow hydrology, sediment supply, riparian functions, channel-floodplain interactions, habitat isolated from salmon access, and water quality. Using a series of diagnostic screens, we locate disturbances to habitat-forming processes, and identify actions (i.e., projects) required to correct the disturbances. This paper reports the progress made by the Skagit Watershed Council in applying its strategy within the range of anadromous salmonids of the Skagit River basin in northwest Washington.

Study Area

The Skagit River Basin drains approximately 8,544 km^2 of the North Cascade Mountains of Washington State and British Columbia (Figure B-2, section A). Elevations in the basin range from sea level to about 3,275 m (10,775 ft) on Mt. Baker. Numerous peaks in the basin exceed 2,500 m in elevation. Average annual rainfall ranges from about 90 cm (35 in) at Mt. Vernon on the lower floodplain, to over 460 cm (180 in) at higher elevations in the vicinity of Glacier Peak. Several vegetation zones occur in the study area. As defined in Franklin and Dyrness (1973), most of the lower elevations are in the western hemlock zone. These forest zones typically include western hemlock (*Tsuga heterophylla*), Douglas fir (*Pseudotsuga menziesii*), western red cedar (*Thuja plicata*), and sitka spruce (*Picea sitchensis*). Deciduous species in this zone include red alder (*Alnus rubra*), black cottonwood (*Populus trichocarpa*), and big leaf maple (*Acer macrophyllum*). Middle elevations are in the Pacific silver fir (*Abies amabilis*) zone, and higher elevations are in the alpine fir (*A. lasiocarpa*) zone (Franklin and Dyrness 1973).

The Skagit River Basin is comprised primarily of mountain drainages, with fewer lowland subbasins (low topographic relief and low elevation). The hydrographs of most low-elevation forested subbasins are dominated by autumn and winter rainfall floods (Beechie 1992). Conversely, spring snowmelt floods typically dominate the hydrographs of high elevation subbasins in the eastern Skagit. Most areas of the Skagit Basin are of intermediate elevation and exhibit both rainfall and snowmelt floods. Lowland subbasins are generally located in the western valley (rain dominated) and are usually more highly developed by urban and agricultural land use than the forested mountain basins (Lunetta et al. 1997).

Land development (primarily logging and draining or clearing lands for agriculture) began around 1860. About 1590 km^2 (615 mi^2, 19%) of the basin is currently in private and State of Washington ownership. Land uses are dominantly agricultural and urban in the lower floodplain and delta areas, and upland areas are generally commercial forests. About 3680 km^2 (1420 mi^2, 44%) of the basin lies within the federally-owned North Cascades National Park, Mt. Baker and Ross Lake National Recreation Areas, and Glacier Peak Wilderness Area; the U.S.D.A. Forest Service controls an additional 1960 km^2 (755 mi^2, 24%) of the basin in the Mt Baker-Snoqualmie National Forest. Approximately 1040 km^2 (400 mi^2, 13%) of the basin is in the Province of British Columbia.

Anadromous salmonids indigenous to the basin include chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), pink salmon (*O. gorbuscha*), chum salmon (*O. keta*), sockeye salmon (*O. nerka*), steelhead trout (*O. mykiss*), cutthroat trout (*O. clarkii*), and native char (*Salvelinus* sp.). Access to anadromous fishes is generally confined to elevations below 700 m by natural barriers. Upstream migration to the Baker River system has been eliminated by the installation of two hydroelectric dams, but anadromous fish production - primarily coho and sockeye salmon - is maintained through trapping and hauling operations, in addition to the maintenance of sockeye spawning beaches and smolt bypass trapping. The extent of salmon upstream migration in the Skagit River Basin is shown in Figure B-2, section B.

Methods

We analyze disturbances to watershed processes in the Skagit River Basin in two phases. In the first (interim) phase, we locate disturbed habitat-forming processes using a combination of existing Geographic Information System (GIS) data and field-based inventories to identify disturbances to peak flows, sediment supply, riparian functions, channel-floodplain interactions, blockages to salmon migration, and water quality. The second phase (to be largely completed during the next two years) relies solely on field-based inventories. Both phases rely on GIS to analyze and maintain landscape process data over the 8,544 km^2 area of the Skagit River Basin. This paper describes only the methods and results from Phase 1.

We have used more than 30 different GIS themes and partial field inventories to apply the landscape process screens identified in the Strategy. The existing GIS themes provide low-resolution data covering the entire river basin. These data give us a good overview of habitat-forming processes in the entire basin, but can give erroneous answers to our questions about specific reach level sites (10^2-10^4 meters linear scale). Field inventories provide high-resolution data, but with only limited coverage at present. Because field-inventories are more reliable at specific sites, the Skagit Watershed Council members have made a long-term commitment to collecting field-based information basin-wide.

We analyzed selected landscape processes that form salmonid habitats in the Skagit River Basin. We selected these analyses based on current scientific knowledge of their effects on salmonid habitat and survival of salmon in freshwater, as well as knowledge of how various land use practices affect the processes (Table B-1). We recognize that the list may not include all impacts to salmon in the watershed. However it includes those that are clearly supported by scientific literature and are responsible for a significant proportion of the total loss in salmon production from the basin. For each process we developed a series of diagnostics based on rates derived from scientific literature and local studies. The diagnostics and methods for estimating values are summarized in Table B-1.

We synthesized the ratings for individual landscape processes and functions into a single reach classification that we call the generalized habitat types. The importance of identifying generalized habitat types for watershed restoration is illustrated by Frissell (1993) and Doppelt et al. (1993), where examples of habitat types are listed along with their biotic objective and restoration tactics. To apply this concept in the Skagit, we derived generalized habitat types based on simple correlations between our understanding of anadromous fish life history strategies and reach level habitat types (approximately 10^2 to 10^4 meters in linear scale) (Table B-2). We assume that relationships between fish life stages and habitat for each indicator species analyzed adequately identifies the "habitats to which salmon stocks are adapted" in an effort to be consistent with our goal.

Our analysis used five species and four life stages to determine generalized habitat types. The life history stages examined were 1) spawning/egg to fry, 2) summer rearing, 3) winter rearing, and 4) estuary rearing. Several salmonid species present in the Skagit River Basin were excluded from the evaluation due to lack of data or a spatial bias in their distribution not related to geomorphic habitat types. Native char were excluded due to a lack of data describing their habitat preferences over their complete life history. We know the spawning range of native char is bias toward higher elevation headwater tributary basins which is included in the range of historical anadromous fish access (Figure B-1, section B). Cutthroat trout were excluded because of their spatial bias towards the lower elevation rain-dominated subbasins of the Skagit. Coho salmon habitat preference is similar cutthroat, and the coho range includes all of the anadromous cutthroat range in the Skagit, therefore we assume that coho relationships in our analysis adequately represent cutthroat. Sockeye were excluded because the population is limited to one subbasin of the Skagit: the Baker River. While resident rainbow trout (*O. mykiss*) are found throughout the entire river basin, they are assumed to have the same juvenile habitat preferences as steelhead (the anadromous form of *O. mykiss*), which are included in our analysis.

Under pristine habitat conditions (i.e., natural disturbances only) we define reach-level habitat types for anadromous salmonids in the Skagit as either “key” or “secondary” (Table B-2). Key habitat is “critical” for at least one life stage or is a “preferred” habitat type by the majority of life stages considered. Secondary habitat does not provide critical habitat for any life stage and is not a preferred type by the majority of life stages considered. Classification systems described in Hayman et al. (1996), Montgomery and Buffington et al. (1997), Peterson and Reid (1984), and Simenstad (1983) were used to define the different reach level habitat types. Local studies used to designate whether the specific reach level habitat types were “critical”, “key”, or “secondary” for a life history stage included: Beamer and Henderson (1998), Beechie et al. (1994), Congleton (1978), Congleton et al. (1981), Hayman et al. (1996), Montgomery et al. (1999), and Phillips et al. (1980, 1981). Data from outside the Skagit (Queets River, Washington, in Sedell et al. 1984) were also used to help understand juvenile fish use differences between large main channels and off-channel habitats.

Under disturbed habitat conditions (i.e., both human and natural disturbances) we designated reach-level habitat types as: “key” when all landscape screening results were rated as functioning; “important” when at least one landscape screen is moderately impaired; “degraded” when at least one landscape screen is impaired; “secondary” when channel type is step-pool or steeper; “isolated” when upstream of a manmade barrier to fish migration, or “unknown.” Some reaches are designated as “unknown” because of the high probability of error in rating the riparian condition correctly by land cover types.

Mainstem areas with any of the following conditions were consider degraded: riparian buffer is less than 20 m wide, streambank edge is hardened (e.g., riprap), or levee is present within 60 m of the bankfull channel edge. All other Lower Skagit mainstem areas were considered important. In the estuary, areas that are hydromodified are considered degraded, areas adjacent to levees are considered important, areas at least one distributary channel away from a levee are considered key.

Results and Discussion

Hydrology – Changes in Peak Flow

We estimate 23% of the mountain subbasins in the Skagit are very likely impaired or likely impaired with respect to peak flow hydrology (Figure B-3, section A). In lowland basins, we estimate 7% of the streams historically accessible to anadromous salmon will be impaired when urban and residential areas are fully built out, and 18% will be moderately impaired (Figure B-3, section B). We use the results shown in Figure B-3 to help evaluate the likelihood of success of proposed restoration projects. In general, we do not support restoration efforts directly in or adjacent to channels that are classified as "impaired" without evidence demonstrating that the proposed work will not fail biologically or physically due to the likely increase to peak flows. We also use the results to identify areas currently in good condition that are planned for future development to an extent that hydrology is likely impaired. For these areas we consider protection actions such as rezoning to a less intensive land use or acquisition. We also identify areas to investigate for potential restoration of hydrologic processes.

Reduced peak flows as a result of flood control change a channel's ability to create and maintain the suite of diverse floodplain habitats to which aquatic species are adapted (Ward and Stanford 1995). Annual peak flows in the Skagit River Basin have changed since flow regulation through the construction of reservoirs capable of flood storage. Before flood storage capability, floods in the lower Skagit River commonly approached or exceeded 5,500 cms, and floods in water years 1815 and 1856 were estimated at 11,327 and 8,495 cms, respectively. Since the advent of flood storage capability, a flood approaching 5,500 cms has not yet occurred. The number of floods between the 2-year and 100-year return period has been reduced by roughly 50% since dams were built on the Skagit and Baker Rivers (Table B-3). Flood storage on the Skagit has likely impacted channel-floodplain processes in reaches downstream of the dams, but we have not yet quantified the effects. Until we have a better understanding of these impacts, we view the dams as watershed controls (i.e., as shown in Figure B-1). That is, they operate independently of our management control because they are licensed for up to 50 years and are unlikely to be removed. Accepting that this disturbance will not likely be altered during the license period of each dam, the artificial creation of off-channel habitat then may be justified in stream reaches where off-channel habitat has been lost due to this disturbance. Alternatively, it may be possible to re-establish certain channel-forming flows that have been eliminated in the past.

Sediment Supply

We estimated that 46% of the area in mountain subbasins of the Skagit is impaired with respect to sediment supply (Figure B-4, section A). Our evaluation of the accuracy of the method showed that it correctly estimated the sediment supply rating for seven of the ten subbasins where sediment budget data were available. It over-estimated average sediment supply for two of the ten test basins (i.e., rated them impaired when they are functioning), and under-estimated sediment supply for one subbasin. Therefore, we recognize that this product should not be used to identify potential restoration projects. Rather, it is used for project screening where field-based sediment budgets are not available, and for general planning watershed-level sediment reduction projects. For project screening, project proponents use this map to determine whether the proposed project area is likely to have an impaired (i.e., high) sediment supply. For reaches where sediment supply is impaired, 1) sediment supply in the watershed should be restored to “functioning” levels before downstream reaches are worked on, or 2) evidence demonstrating that the proposed work will not fail due to increased sediment supply should be presented.

Specific sediment reduction projects are identified based on the results of forest road inventories. We focus on forest roads for sediment reduction projects because mass wasting rates from forest roads averaged about 44 times more than mass wasting rates in mature forest (Paulson 1997). Currently, about 1,300 km of road are inventoried with another 3,000 km remaining (Figure B-4, section A). Risk ratings from the current inventory showed that a significant number of forest roads in the Skagit Basin pose a landslide hazard and potentially threaten fish habitat. Based on this inventory, we will focus initial sediment reduction projects on the high-risk and moderate-risk road segments.

For example, the Bacon Creek watershed has 3.7 km and 18.6 km of high and moderate risk roads respectively (Figure B-4, section B). The high-risk road segments cross relatively more of the landforms sensitive to disturbance for mass wasting (e.g., inner gorges and steeper hillslopes) than moderate or low risk roads. Sediment reduction projects on these roads would reduce the risk of sediment supply increasing due to roads therefore increase the level of watershed protection. Specific road projects primarily involve stabilizing or re-contouring road fills, removing stream crossings or improving drainage conveyance, and improving road surface drainage. For river basin level planning, we consider subbasins with the lowest total cost of road restoration per kilometer of salmon stream as the highest priority.

Riparian Function

Before interpreting the Landsat classification of riparian forests, we used field inventory results from 234 riparian sites to describe the distribution of field-based riparian classifications within each satellite-based forest class (Table B-4). All of the sampled “late-seral forest” sites, and between 92% and 88% of the “mid-seral forest” and “early-seral forest” sites met the > 40 m wide riparian buffer criteria, fitting our functioning designation. Conversely, 90% of the areas mapped as “non-forest” had < 20 m wide riparian buffers, fitting our impaired designation. Areas mapped as “other forest” (ranging from clear-cuts to mature hardwoods) were found to be 43% functioning, 15% moderately impaired, and 42% impaired.

Based on this analysis, we estimate that 29% of the non-mainstem channels in the anadromous zone (by length) are in the non-forest land cover category, and therefore have a very high likelihood of being impaired and in need of riparian restoration (Figure B-5). Conversely, 19% of the non-mainstem channels in the anadromous zone are in the mid to late seral forest land cover category, and therefore have a high likelihood of being functioning and therefore needing protection. While we cannot accurately map stream reach scale riparian conditions associated with channels adjacent to all GIS land cover types, we can estimate with reasonable accuracy the total of each riparian category at a larger scale. Based on the results in Table B-4, we estimate the percentage of non-mainstem channel length in the Skagit anadromous zone by each land use designation (Figure B-6).

We rely on field inventories to identify actual restoration projects because of the above-mentioned limitations in the satellite classification of riparian forests. We conducted field inventories of riparian forests by walking all streams accessible to anadromous fish and assessing the riparian vegetation conditions for each stream reach. We classified riparian conditions by buffer width, stand type, and age of vegetation within 60 m of stream channels. From these data, we selected all stream segments with forested riparian vegetation less than 40 m wide as requiring planting, and all segments with evidence of livestock access to the stream channel as requiring fencing. Riparian planting and restoration projects have been identified through a series of field inventories. The inventories were completed systematically as four separate projects between 1995 and 1998 in 24% of the Skagit's subbasins (Figure B-5). Together, the inventories identified 130 km of stream corridor for riparian planting and fencing projects.

Isolated Habitats and Disrupted Channel-Floodplain Interactions

The inventory efforts through September 1999 have identified 229 manmade barriers out of 572 channel crossing structures with 32% of the anadromous zone inventoried. In tributary habitat, 143 km of channel is blocked. In the delta, we estimate 185 km (56%) of the channels have been isolated or lost to salmon access under present conditions (Figure B-7). Isolated channels are those where a topographic channel and water exist, but juvenile or adult salmon access is blocked due to man-made disturbances. Lost channels are those areas that were historically channels, but currently do not have clear a topographic channel or water present.

Because manmade barriers are not evenly distributed throughout the Skagit River Basin and our inventory efforts have focused in areas where barriers are more common, we anticipate that the majority of the isolated habitat in the basin has been found. Based on a subsample of 111 inventoried structures within subbasins of the Skagit with similar land-use intensity as the subbasins yet to be inventoried, we found that 14% of the inventoried structures do not meet fish passage criteria. Therefore, we expect to find about 150 more blockages in non-inventoried areas of the basin, blocking about 60 km (4%) of the estimated length of tributary habitat in the anadromous zone.

Upstream of the Skagit delta, 46 km of stream banks have been riprapped (Figure B-7). In the geomorphic delta, 51 km (62%) of the mainstem channel edge is either hardened, diked within 60 m of the channel's edge, or both. These inventory results provide the basis for identifying potential riprap removal (or modification) projects, primarily where hardened banks no longer protect capital improvements (e.g., house, road).

Generalized Habitat Types

The final result of our analysis is the identification of generalized aquatic habitat types throughout the entire river basin, which are based on salmonid habitat preferences combined with the results of the landscape process screens. The resulting analysis in the Skagit Basin yields a mosaic of reach-level habitat patches (Figure B-8). Key habitat areas have all habitat-forming processes functioning at or near historic levels, and are targeted for habitat protection. Because protection of habitat is generally considered less expensive than restoration, we view key habitats as some of the highest priority areas for habitat expenditures. Isolated habitats are typically the most cost-effective restoration projects, and therefore receive strong consideration for restoration funding. Important and degraded habitats are both areas that are targeted for restoration.

Secondary habitat will not be targeted for restoration under this strategy. That is, we do not intend to "restore" secondary habitat to key habitat. However, it is important to understand how secondary habitat may function in a watershed in order to protect or restore the other habitat types. For example, the source of degradation may originate in secondary habitat (i.e., the idea of contributing critical areas, discussed in Frissell, 1993). In such cases, restoration of processes originating in secondary habitat areas may be required in order to restore downstream degraded or important habitats.

Identification of Restoration Projects

The main objective of the strategy is to identify habitat protection and restoration projects based on application of the landscape process screens. Together, our analyses have led to the identification of more than 400 individual restoration projects. For example, our analysis of the U. S. Forest Service road inventory identified approximately 650 km of high-hazard and moderate-hazard roads that are candidates for restoration. The total estimated cost for all of these roads (which does not include forest roads on state and private lands) is approximately $11.6 million. We also identified 122 riparian planting and fencing projects during inventories of only 24% of the river basin, with a total cost estimated at $1,687,000. Of these riparian projects, 39 are already funded.

We completed migration barrier inventories in 13 out of 38 subbasins, and identified 229 blocking structures. Some blockage removal projects have uncomplicated designs and relatively clear benefits. These projects can each be considered independently of other culverts because salmon currently access the culvert sites, and repair of the structures will provide benefits commensurate with the amount of habitat upstream. By contrast, groups of completely blocking structures on the same watercourse should be considered either in combination or sequentially, and projects that involve flood protection levees or coordination of numerous landowners require feasibility studies to determine suitable restoration actions. Currently we have a list of 36 isolated sloughs and blind tidal channels that require further assessment for design of appropriate solutions.

Each restoration project is mapped on a GIS theme, and relevant data are stored in the associated databases. These themes can be updated as new inventories are completed, or as project status changes (e.g., design phase, construction, completed). Additionally, we can develop related databases for monitoring the effectiveness and costs of different project types. Over time, the GIS maps and databases will help display progress made in restoring habitats in the Skagit River Basin, and will help us modify our actions to more efficiently restore habitat in the basin.

Current Limitations and Future Work

Both lowland and mountain basin GIS-based results give us operating hypotheses for peak flow impairment throughout the river basin. Peak flow ratings for mountain subbasins in the Skagit were developed based on an empirical correlation between land use and elevated peak flows an adjacent basin (North Fork Stillaguamish River) because subbasin flow data are limited in the Skagit. The North Fork Stillaguamish River has exhibited a 38% increase in mean annual maximum flow between 1928 and 1995 with climatic variables explaining less than 40% of the increase suggesting that changes in the watershed condition has caused the balance of the increase (Beamer and Pess 1999). However, future efforts for the mountain basin methodology must confirm that correlations between land use and peak flows in the North Fork Stillaguamish are a cause-effect relationship, and then identify the appropriate thresholds for land use before re-applying to the Skagit. The lowland basin methodology should be repeated with land cover data that estimates current effective impervious area to complement the results reflecting impervious area at fully developed watershed conditions per zoning designations.

Field-based sediment budgets more accurately estimate the sediment supply in a subbasin, and describe the relative effects of different land uses on sediment supply. Therefore, they will provide more accurate information for project screening and planning than do our current GIS-based estimates. Field-based sediment budgets have been completed for approximately 12% of the total area (Paulson 1997), and sediment budgets for the remainder of the basin will be completed in 2001. In lowland basins, mass wasting is not a dominant sediment supply process, but increased fine sediment supply to channels is directly related to urban, livestock grazing, and agricultural land use. We anticipate future development of a surface erosion and sedimentation screen for these low-slope areas focusing on quantifying surface erosion from agricultural or developed areas.

The U.S. Forest Service is continuing its road inventory. Similar road inventories have not yet been conducted on state or private timberlands. The inventory method appears to be appropriate for identifying segments of road that pose the greatest threat to stream resources. However, it does not identify the types and locations of work needed to reduce the landslide hazard. We anticipate that some inventories will be more detailed than those used by U.S. Forest Service, and will better identify the specific work actions required for each segment of forest road.

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Satellite data do not provide sufficient information for identifying all riparian restoration and protection actions at the stream reach level. The GIS-based riparian screen is reliable for only late and mid-seral conifer dominated forest and non-forest areas. Because of the higher probability of error in rating stream reaches by the remaining land cover types, they are excluded from the interim screen. Therefore, the interim riparian screen is applied to only about 50% of the anadromous zone (based on length). Field inventories are far more reliable than remote sensing data, and can provide sufficient information for stream reach level project planning. Therefore our primary task is to complete field inventory of riparian forest conditions throughout the river basin.

The field-based inventory of man-made blockages to salmon migration has been completed for only a portion of the basin, but the future inventories are fully funded and should be completed by 2001. Areas currently identified as "isolated" are accurately characterized as upstream of man-made blockages to salmon because they are based entirely on field inventory. We assume that some areas yet to be inventoried are "isolated", although extrapolation from current inventories suggests that no more than 4% of the remaining channel length is likely to be upstream of a man-made blockage to salmon migration. In addition to the remaining blockage inventories, we have yet to complete our inventory of wetland habitat losses in the delta. The wetland inventory should also be completed by 2001.

Water quality parameters such as dissolved oxygen, temperature, turbidity, nutrient loading, and levels of toxic substances are critical to salmon health and survival. Identifying areas where water quality is impaired and the various factors contributing to impairment, and then addressing the causes of water quality degradation is important to restoring salmon habitat in the basin. Currently, we consider stream reaches, lakes, and estuary areas that are included on the Washington Department of Ecology's Candidate 1998 Section 303(d) Impaired and Threatened Water Bodies listings as "impaired." These water bodies are known to fail Washington State's surface water quality standards, and are not expected to improve within the next two years. We anticipate our future water quality screen to include locations of known point and non-point sources that may contribute to water quality degradation in the basin. These land use indicators will be used to identify areas where water quality problems may exist, and to direct further investigation (e.g., water quality sampling, benthic invertebrate community analyses) to determine if water quality is actually impaired. The continuing objective will be to improve the quality and quantity of water quality data and land use information available to guide restoration and protection of aquatic habitats.

The primary limitations in accurately identifying generalized habitat types are incomplete natural landscape process screens and the accuracy of individual screens used. The consequence of incomplete landscape process screens is an underestimate in the amount of "degraded" and "important" habitat, and an overestimate of the amount of "key" habitat. However, we have high confidence that areas identified as "degraded" are in fact degraded. That is, there is a very low likelihood that areas identified as "degraded" with this analysis will later be identified as "important" or "key" habitat. Conversely, some areas identified as "key" habitat with this analysis will be changed to "degraded" or "important" as more detailed information becomes available.

Conclusions

The Skagit Watershed Council first identified its conceptual framework and diagnostic criteria, thus enabling systematic application of a strategy supported by all members. Without this step, a systematic and objective inventory of habitat problems in the Skagit Basin would not have been possible. Following development of the strategy, the Council quickly applied the simplest diagnostic criteria over the entire basin with limited funds. This effort yielded many “no-brainer” project ideas (some are socially “risky”), as well as a good overview of the spatial pattern of disturbance over the entire basin. By contrast, a haphazard inventory or professional judgment system would have produced lists of projects, but would not necessarily give managers the tools to be strategic or comprehensive in restoring salmon habitats. In other words, managers would be unable to focus on important biological hotspots or impaired landscape processes because they lacked a comprehensive understanding of the causes of habitat degradation in the basin.

The strategy recognizes that land use and resource management activities influence natural landscape processes, which result in changed habitat conditions (Figure B-1). Therefore, restoration and protection actions identified by implementing this strategy should be directed at the habitat-forming processes instead of attempting to build specific habitat conditions. Focusing actions on “building” habitat for specific species may be to the detriment of other species and may not be sustainable due to potential conflicts with natural processes (Frissell and Nawa 1992, Kauffman et al. 1997, Beechie and Bolton 1999). Instead, actions implemented by this strategy will aim to create the conditions necessary for natural landscape processes to reestablish at levels similar to those that existed historically which should: 1) result in a high likelihood of long-term project success, 2) protect and restore habitat for all salmonid species as well as other native aquatic and riparian dependent species, and 3) ensure the effective use of public and private restoration funds.

The Skagit Watershed Council overcame diverse interests to develop and apply the interim phase of its restoration and protection strategy in about a two-year period. The field inventory phase of the strategy will be completed during the next two years. The cost of all inventories and analyses required to develop a restoration plan (including a list of required restoration and protection actions) for the 8,500 km^2 basin is only about $1.1 million. This total is less than the cost of opening one large isolated estuary channel and wetland complex ($1.9 million - U.S. Army Corps of Engineers, 1998) or a few culvert blockages on local or state highways ($250,000 to $600,000 each – Skagit County Public Works). The estimated cost of potential projects identified during this first phase of applying the strategy is well over $100 million, suggesting that the total cost of the inventories will be less than 1% of the cost of restoration and protection actions. Moreover, development of the restoration plan should save millions of dollars by avoiding projects that are not effective at restoring salmon habitats. On a per unit area basis, the cost of all interim assessments and final field inventories will total only $210 per km^2 assuming that costs remain relatively constant during the next two years.

Application of the strategy gives the Council the ability to become truly strategic in their restoration and protection efforts by providing a consistent set of principles that guide restoration actions, and by systematically identifying hundreds of restoration and protection projects that can be prioritized and sequenced logically. Having a complete river basin overview of landscape processes and resulting habitat conditions allows the Council to set goals on how much restoration or protection is needed to meet a specific priority. The strategy allows priorities to be based on locally defined objectives such as recovery of a certain species or completion of certain types of restoration (Lichatowich et al. 1995, Beechie et al. 1996). However, prioritization does not alter the types of projects enacted, but only alters the sequence in which projects are completed (Beechie and Bolton 1999). Currently the Skagit Watershed Council prioritizes projects based on the relative cost-effectiveness of different projects, which means that projects protecting or restoring the greatest proportion of anadromous fish habitat function per dollar cost are considered higher priority. Additionally, individual restoration groups may choose projects from any list of projects in order to fulfill their respective missions.

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Table B-1. Summary of background and methods used for rating individual landscape processes.

Background	Method description	References
Hydrology – peak flow in lowland basins		
The degree of urbanization is correlated with increased flooding, degraded habitat, and lower salmonid populations in lowland basins of Puget Sound. When the 2-year flood magnitude under current land use equaled or exceeded the 10-year flood under "forested" watershed conditions, channels were consistently degraded. Watersheds with effective impervious area (EIA) greater than 10% always had degraded channels whereas watersheds with EIA ≤ 3% had high species and habitat diversity and abundance.	Hydrologic impairment in lowland basins rated based on planned effective impervious area (EIA), which is the weighted average EIA upstream of the stream reach under fully developed conditions per land use zoning designation. Weighted average EIA was calculated using GIS by assigning EIA values to polygons based on land use zoning designations. EIA ≤ 3% is considered "functioning", EIA between 3% and 10% is "moderately impaired", and EIA >10% is "impaired."	Beyerlein (1996), Booth and Jackson (1997), Dinicola (1989), Moscrip and Montgomery (1997), WDF (1994)
Hydrology - peak flow in mountain basins		
Increased peak flows result in an increased frequency of channel forming and bed mobilizing flow events, leading to channel destabilization, less complex habitat, and increased bed scour depths significantly affecting salmonids. Two common land use causes of increased peak flow in forested mountain basins relate to hydrologically immature vegetation and forest road drainage. Hydrologically immature vegetation has relatively low canopy density in winter, allowing increased snow accumulation and melt, resulting in higher runoff rates than areas with hydrologically mature vegetation. Forest road ditches extend the channel network, resulting in more rapid routing of water to main stream channels than basins without road networks.	Peak flow ratings for mountain subbasins in the Skagit were developed based on an empirical correlation between land use and elevated peak flow in an adjacent basin because subbasin flow data are limited in the Skagit. Subbasins with more than 50% watershed area in hydrologically immature vegetation due to land-use and more than 2 km of road length per km^2 of watershed area are rated "very likely impaired". Subbasins exceeding one or the other of the criteria are considered "likely impaired". Subbasins that do not exceed either criterion are considered "functioning."	Beamer and Pess (1999), Jones and Grant (1996), Lunetta et al (1997), Montgomery (1993), Washington Forest Practices Board (1995)
Sediment		
Clear-cutting and forest roads increase landsliding and the supply of coarse sediment (>2 mm diameter) to stream channels, although fine sediments (< 2 mm diameter) are also	Estimating impairment of sediment supply: Average sediment supply for each subbasin estimated based on average sediment supply rates for 13 combinations of geology and vegetation	Collins et al. (1994), Dietrich et al. (1989),

delivered by mass wasting. Large increases in coarse sediment supply tend to fill pools and aggrade the channel, resulting in reduced habitat complexity and reduced rearing capacity for some salmonids. Large increases in total sediment supply to a channel also tend to increase the proportion of fine sediments in the bed, which may reduce the survival of incubating eggs in the gravel and change benthic invertebrate production. Landform and land use both influence mass wasting rates. Most sediment from mass wasting originates from inner gorge landforms (steep, stream-adjacent slopes). On average, these areas cover less than 20% of the mountain basins in the Skagit but produce about 75% of the sediment delivered to streams. Hillslopes >30° are also generally unstable tending to produce shallow-rapid landslides from bedrock hollows or channel headwalls. Hillslopes <30° are generally stable. Deep-seated failures, usually located in glacial deposits or phyllite, have high mass wasting and delivery rates to streams. Compared to mature forest, the increase in mass wasting rates for clear-cut forests and forest road areas averages about 6 and 44 times higher, respectively.	cover (Landsat '93), which were derived from nine sediment budgets conducted within the basin. Using GIS we calculated average current sediment supply for each subbasin, and the average increase over the natural sediment supply for each subbasin (current/natural). Sediment supply process is considered "functioning" where average sediment supply is <100 $m^3/km^2/yr$, or where average sediment supply is >100 $m^3/km^2/yr$ but <1.5 times the natural rate. Sediment supply is "impaired" where average sediment supply is >100 $m^3/km^2/yr$ and >1.5 times the natural rate. Forest road inventory - identify sediment reduction projects: The inventory rates factors that influence road-related landslides and the consequences of landslides. All ratings concerning the likelihood of landsliding are summed, and then multiplied by a rating of the likelihood that significant stream resources will be impacted. The final value, called the risk rating, ranks roads with respect the threat that they pose to salmon habitat. Higher risk ratings indicate greater chance that a road will fail and impact salmon habitat. Final ratings were grouped into three categories of risk. A rating >30 is high, 16 to 30 is moderate, and ≤15 is low.	Lisle (1982), Lisle (1989), Lunetta et al. (1997), Madej and Ozaki (1996), Paulson (1997), Peterson et al. (1992), Renison (1998), Sidle et al. (1985)
Riparian Function		
Clearing of riparian forests can alter large woody debris (LWD) recruitment to streams, which in turn alters the habitat characteristics of streams. Reduced LWD recruitment persists for several decades, leading to declining LWD abundance in the first few decades and sustained low LWD abundance between 50 and 100 years after the disturbance. A change in LWD abundance alters fish habitat characteristics such as pool spacing, pool area, and pool depth, and this alteration of habitat characteristics causes changes in the salmonid carrying capacity of a stream.	Remote sensing assessment: Riparian forests that can produce 80% of potential late-seral LWD recruitment over time (> 40 m wide) are considered "functioning." Riparian forests producing 50% to 80% of the potential late-seral recruitment (20 to 40 m wide), are considered "moderately impaired." Buffer widths less than 20 m are considered "impaired." We estimated the proportion of impaired, moderately impaired, and functioning riparian forests by using LANDSAT classifications of vegetation. Field inventory: Ratings are the same as above. In addition to documenting forested buffer width, field inventories also classify stand types by species mix and seral stage, which gives us sufficient information to prescribe generalized	Abbe and Montgomery (1996), Beechie and Sibley (1997), Bilby and Ward (1991), Hicks et al. (1991), Lunetta et al. (1997), Montgomery et al. (1995), Murphy and

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	management regimes for each segment of riparian forest. Inventories also identify areas of livestock access and potential fencing projects.	Koski (1989), Murphy et al. (1985)
Channel-floodplain		
Disconnecting rivers from their floodplains changes the ability of a stream to supply, transport, or store one or more of the inputs: water, sediment, and wood. This constrains the formation and maintenance of habitat within floodplains. Stream bank hardening (hydromodification) prevents channel migration, reduces LWD recruitment, and typically narrows and steepens channels increasing both sediment and water transport rates. Mainstem channels in the Skagit dominated by hydromodification exhibited less diversity in edge habitat types and less edge habitat area than non-hydromodified mainstem reaches. Juvenile chinook and coho salmon abundance was strongly correlated to wood and other natural cover types when compared to riprap or rubble cover, commonly used for stream bank hardening.	Floodplain areas were delineated where the 100-year floodplain was greater than two channel widths using Federal Emergency Management Agency maps or U. S. Geological Survey 7.5-minute quadrangles and aerial photographs. Reach breaks were based on differences in floodplain width and changes in channel pattern. Hydromodified areas were delineated on copies of aerial photos by rafting or jet boating each main channel within floodplain reaches, then digitized into a GIS arc theme.	Beamer and Henderson (1998), Hayman et al. (1996), Ward and Stanford (1995)
Isolated habitat		
Isolation of habitat by levees and culverts has dramatically reduced carrying capacity of the Skagit Basin over the past 150 years. This includes blockages that impede upstream migration of adult salmon seeking suitable spawning areas as well as blockages to other life stages such as juvenile rearing habitat in both the freshwater and estuarine environment. Some isolated habitat can be recovered by simply removing the barrier (e.g., re-building road crossings that block passage), whereas others will require feasibility studies to determine a range of possible alternatives to accommodate both fish use and existing land use.	Man-made barriers to anadromous fish habitat are identified through a systematic field inventory of channel crossing structures (culverts, tidegates, bridges, dams, and other manmade structures). The inventory identifies the type and physical dimensions of structures as well as physical attributes necessary for modeling water flow conditions and comparing results to passage criteria for salmonids.	Beechie et al. (1994), Collins (1998), WDFW (1998)

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Table B-2. Designation of generalized habitat types. Key habitat is critical (i.e., required for the persistence of a dominant life history type) for at least one life stage or is preferred habitat type by the majority of life stages considered. Secondary habitat (sec) does not provide critical habitat for any life stage and is not a preferred type by the majority of life stages considered.

Reach-level habitat type	Chum	Coho	Chinook	Steel-head	Pink	Total number of life stages examined	Percent of all life stages designated "key" or "critical"	Overall designation for "pristine" habitat
<u>Tributaries reaches:</u>								
Ponds (including beaver ponds and other wetlands with significant open water area)	sec	critical	key	key	sec	10	60%	key
Pool riffle	key	key	key	key	key	10	90%	key
Forced pool riffle	sec	key	key	key	key	10	85%	key
Plane bed	sec	sec	sec	sec	sec	10	0%	sec
Step-pool	sec	sec	sec	key	sec	10	15%	sec
Cascade	sec	sec	sec	key	sec	10	15%	sec
<u>Main river reaches:</u>								
Main channel floodplain < 2 channel widths	sec	sec	sec	key	sec	10	15%	sec
Main channel floodplain > 2 channel widths	key	sec	key	key	key	10	80%	key
Off-channel habitat (e.g., ponds, sloughs, side channels, oxbow lakes)	key	critical	key	sec	sec	10	60%	key
<u>Estuary:</u>								
Estuarine or tidally influenced wetlands	key	sec	critical	sec	sec	5	40%	key
Blind channel	key	key	critical	sec	sec	5	60%	key
Subsidiary channel	key	key	key	sec	key	5	80%	key
Main channel	key	key	key	sec	key	5	80%	key

Table B-3. Magnitude of peak flows for the lower Skagit River before and after flood storage capability. Estimates for the period prior to flood storage capability on the Skagit are from a gage near Sedro Woolley (river km 36), reported in Williams et al. (1985). Estimates for the period after flood storage capability are from Sumioka et al., (1998) using data from the gage near Mount Vernon (river km 25).

Flood return period (years)	Before flood storage (cms)	After flood storage (cms)
2	3,147	1,830
5	4,735	2,479
10	5,862	2,934
25	7,361	3,540
50	8,528	4,015
100	9,734	4,508

Table B-4. Distribution of 234 field-sampled riparian sites by GIS-based land cover type.

	Late-seral forest (n=24)	Mid-seral forest (n=13)	Early-seral forest (n=24)	Other forest (n=96)	Non-forest (n=77)
< 20 m forested buffer "impaired"	0%	8%	8%	42%	90%
20-40 m forested buffer "moderately impaired"	0%	0%	4%	15%	6%
> 40 m forested buffer "functioning"	100%	92%	88%	43%	4%

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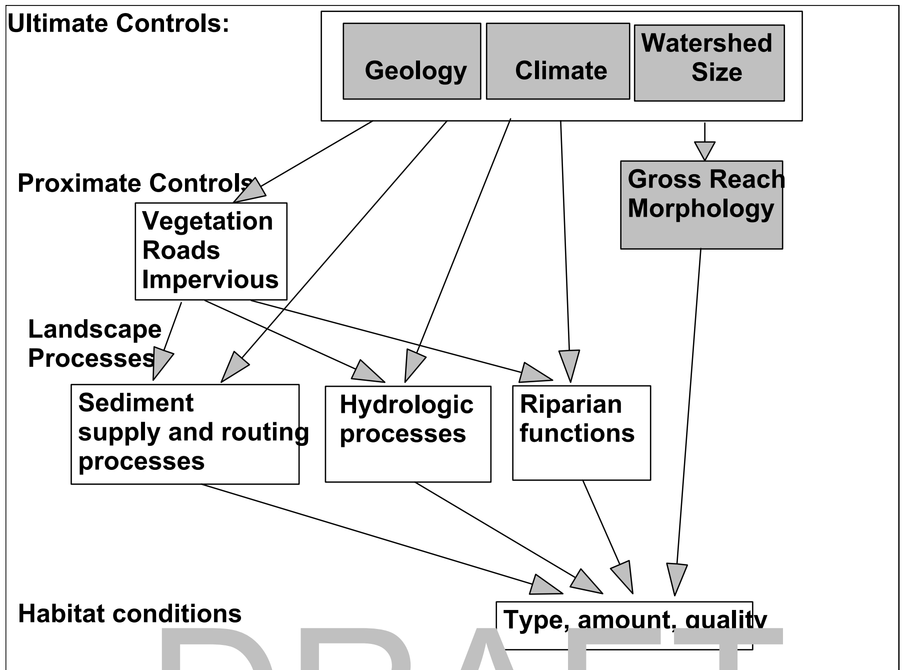

Figure B-1. Simplified flow chart depicting interactions between watershed controls and processes resulting in physical habitat conditions. Shaded boxes represent components that are not influenced by land and resource management while unshaded boxes represent components that are influenced by land and resource management. Pathways for water quality and nutrient cycling are not depicted in this flow chart.

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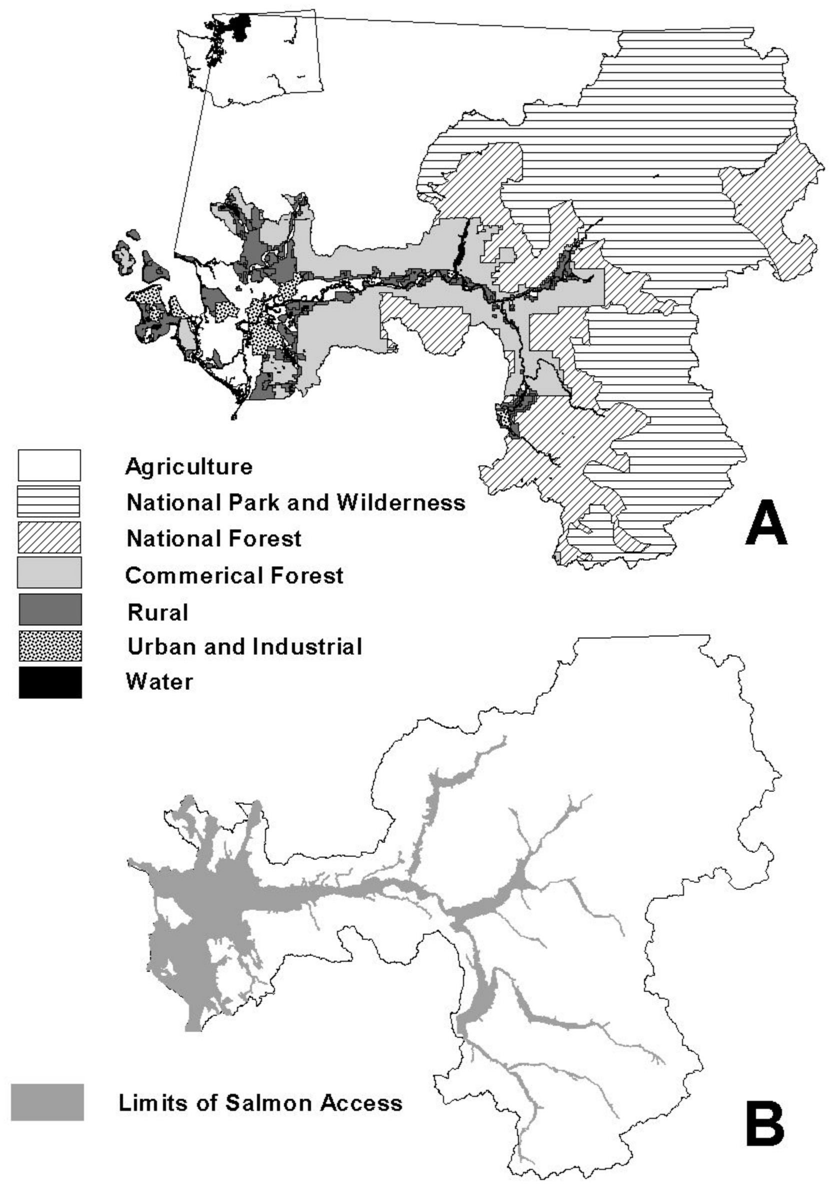

Figure B-2. Location, land use pattern (A), and area of historical salmon access in the Skagit River Basin in Washington State (B).

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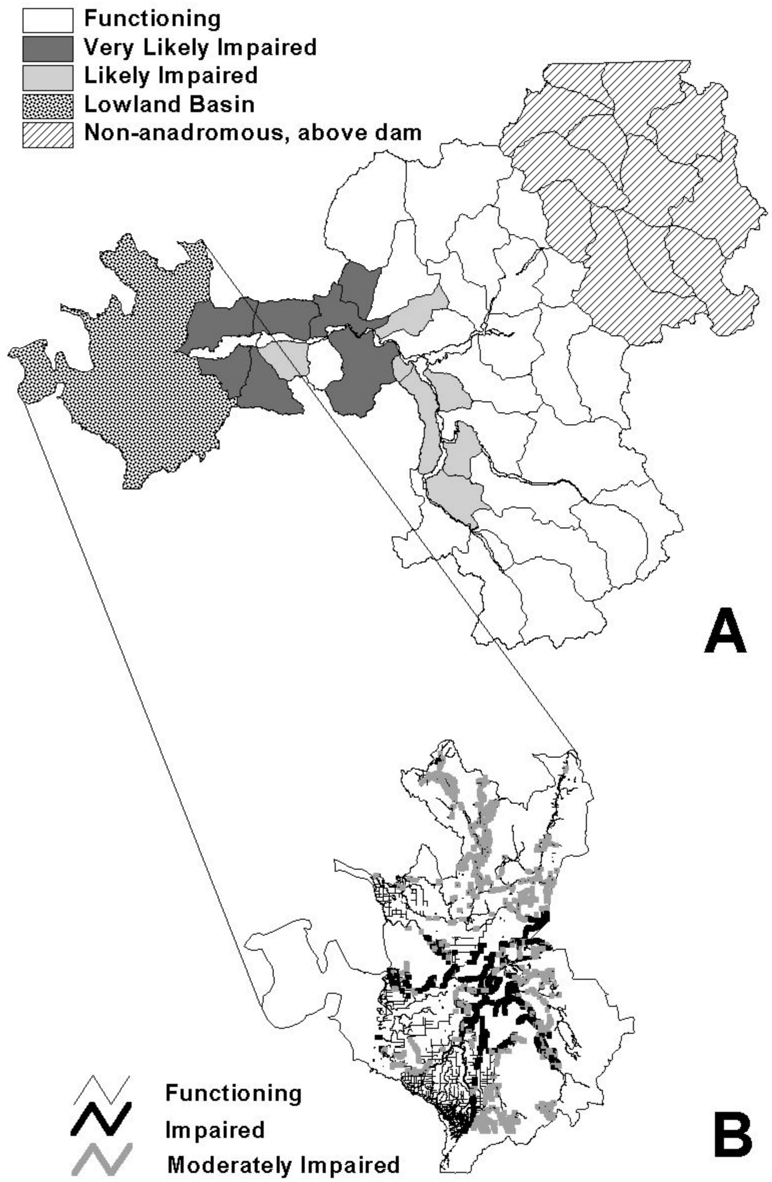

Figure B-3. Subbasins in forested mountain areas of the Skagit River Basin where peak flow is likely impaired (A), and streams in lowland basins where peak flow is planned to be impaired (B).

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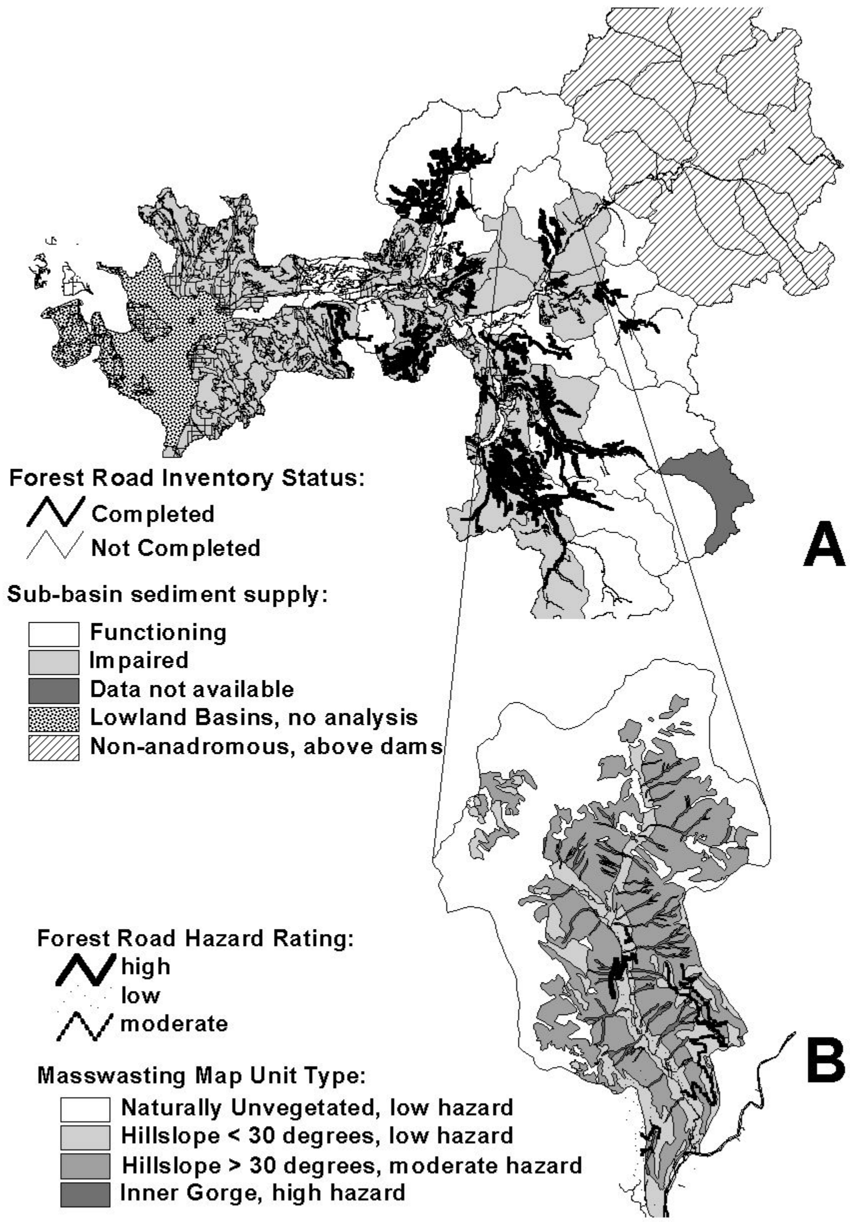

Figure B-4. Sediment supply ratings for mountain basins and status of forest road inventory throughout the Skagit Basin (A), and example of detail for road segments and landslide hazard units in the Bacon Creek watershed (B).

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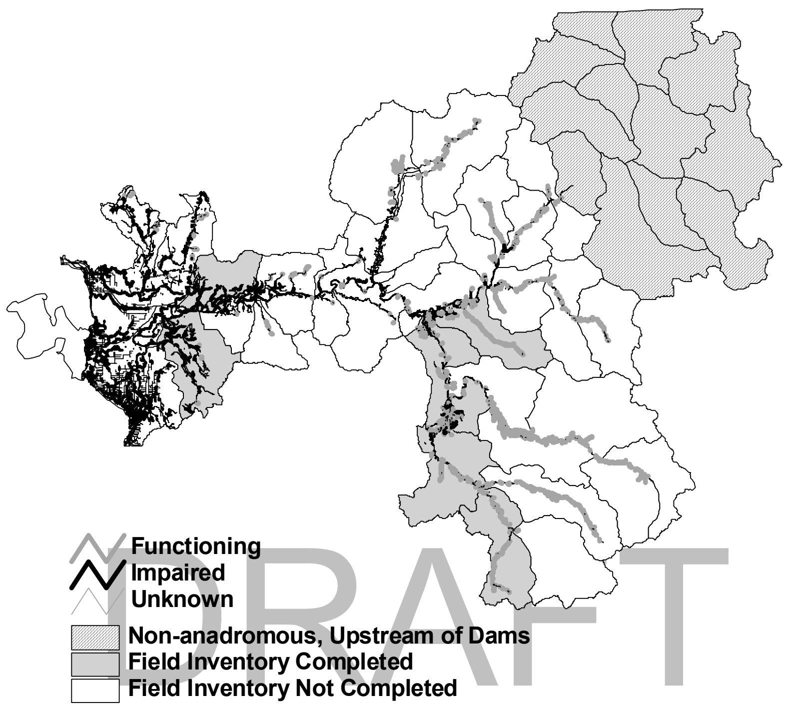

Figure B-5. Map of riparian areas likely impaired or functioning in the Skagit River Basin. Shaded subbasins are where field-based riparian inventories have been completed.

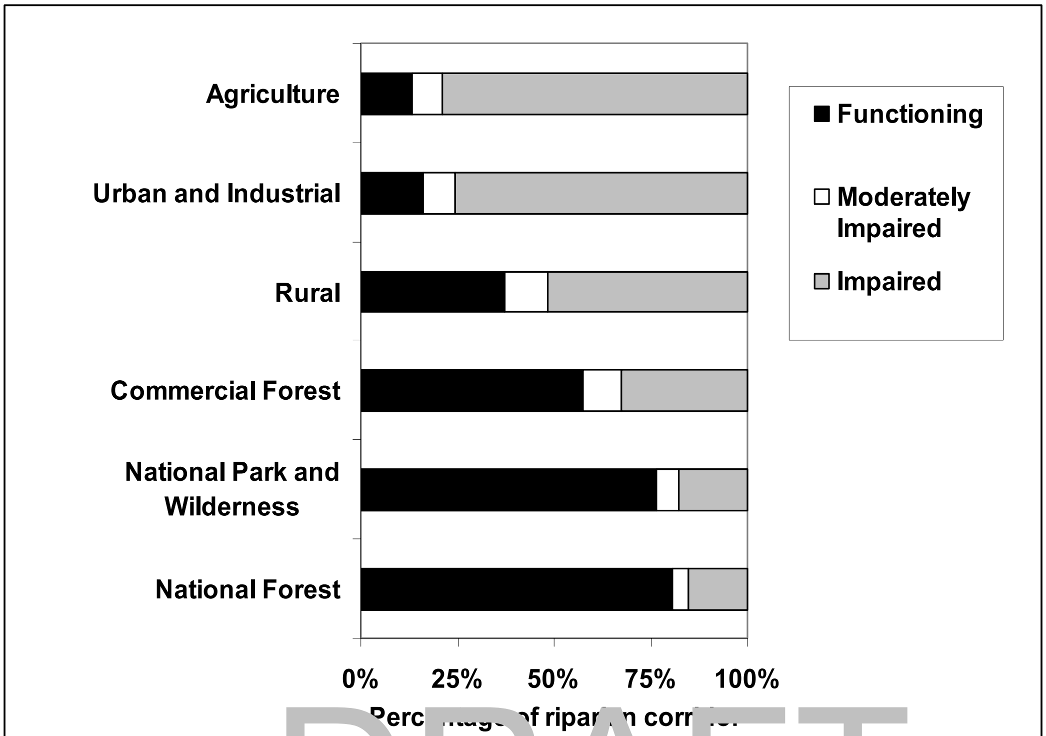

Figure B-6. Estimated percentage of riparian category (impaired, moderately impaired, and functioning) along non-mainstem channels in the anadromous zone of the Skagit River Basin.

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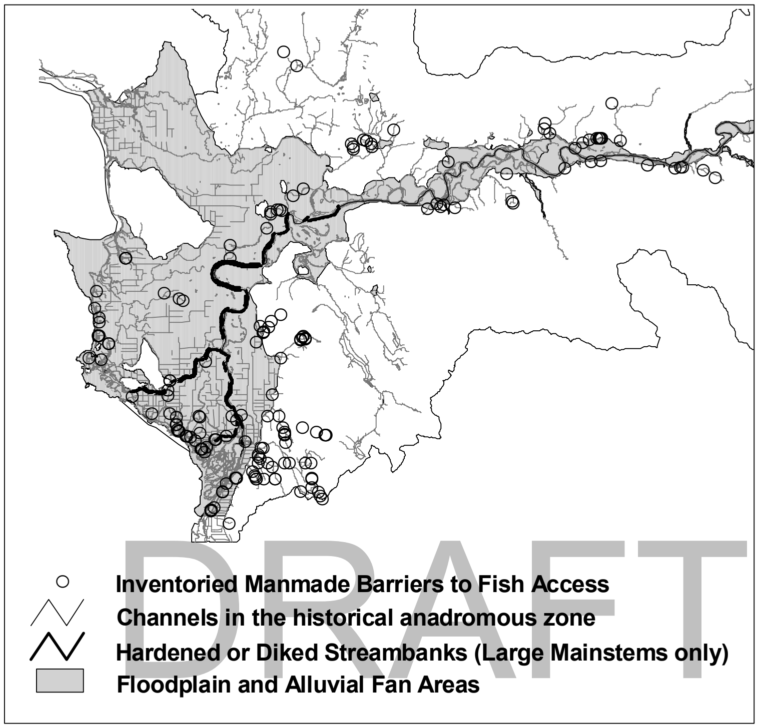

Figure B-7. Location of hydromodification and man-made barriers (Lower Skagit Basin only).

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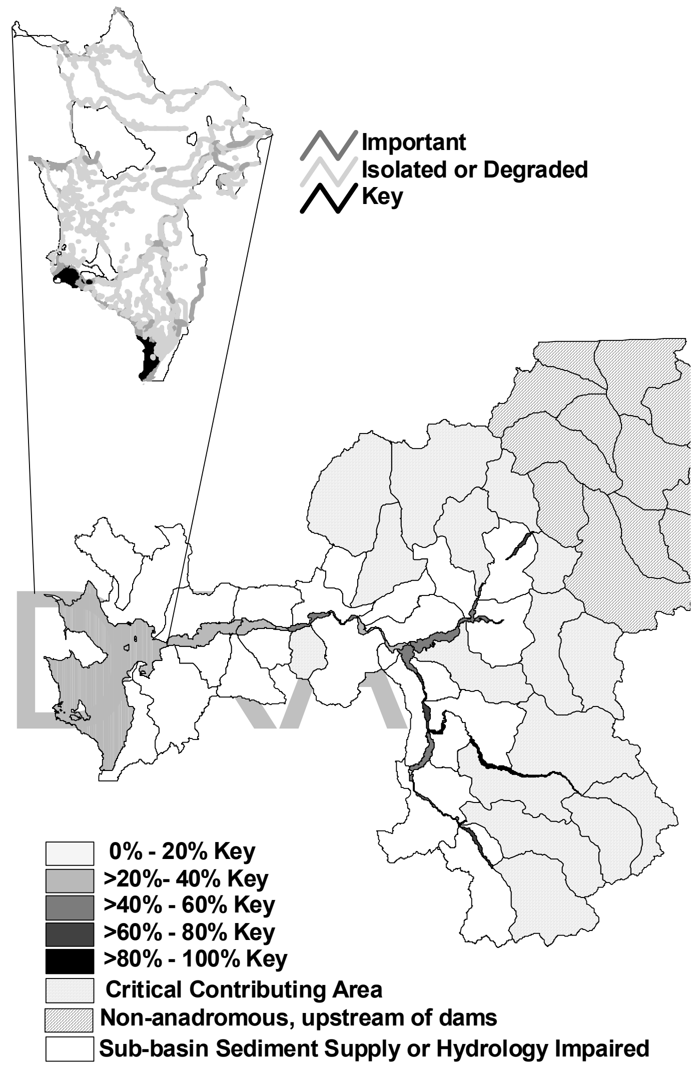

Figure B-8. Distribution of generalized habitat types throughout the Skagit River Basin with example of detail for the delta region.